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ENVIRONMENTAL POLLUTION 14

Wastewater Treatment in Constructed Wetlands with Horizontal Sub-Surface Flow

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ENVIRONMENTAL POLLUTION

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Wastewater Treatment in Constructed Wetlands with Horizontal Sub-Surface Flow

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TABLE OF CONTENTS

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Lenka Kröpfelová received her MSc. degree from the Department of Technology of Silicates at the Prague Institute of Chemical Technology. Between 1995 and 2001, she was affiliated with the environmental company ENVI, in Třeboň, as a specialist on hydrochemistry of freshwaters and natural wetlands. In 2001, she joined NGO ENKI, o.p.s. as an environmental researcher and since 2003 she has been mainly focusing on constructed wetlands for wastewater treatment. Lenka Kröpfelová is a member of the International Water Association and Society of Wetland Scientists.

PREFACE

Wetlands have been used for uncontrolled wastewater disposal for centuries. However, the change in attitude towards wetlands during the 1950s and 1960s caused the minimization of the use of natural wetlands for wastewater treatment (at least in developed countries). Constructed wetlands have been used for wastewater treatment for about forty years. Constructed wetland treatment systems are engineered systems that have been designed and constructed to utilize the natural processes for removal of pollutants. They are designed to take advantage of many of the same processes that occur in natural wetlands, but do so within a more controlled environment.

The aim of this book is to summarize the knowledge on horizontal sub-surface flow constructed wetlands (HF CWs) and objectively evaluate their treatment efficiency under various conditions. The information on this type of wastewater treatment technology is scattered in many publications but a comprehensive summary based on world-wide experience has been lacking. The book provides an extensive overview of this treatment technology around the world, including examples from more than 50 countries and examples of various types of wastewater treated in HF CWs.

As such, the book's intention is to provide a broad base of knowledge, including 1) basic information about processes occurring in wetland soils and overlying water, 2) general information about various types of constructed wetlands for wastewater treatment, 3) detailed information about functioning, performance, operation and maintenance, and costs of sub-surface horizontal flow constructed wetland, 4) information on the use of HF CWs for various types of wastewater around the world, and 5) literature sources dealing with constructed wetlands, especially with HF CWs. The book is not intended as design manual and therefore it does not contain detailed guidelines for construction of these systems. Also, it is not the intention of the authors to provide a detailed theoretical analysis and does not deal with modeling. For this kind of focused practical theory, readers may wish to refer to other books – see *Suggested Reading* at the end of this volume.

Chapter 1 provides a general overview on wetland functions and values, and a brief history about the use of natural and constructed wetlands for wastewater treatment. The second chapter deals with oxidation-reduction conditions and transformations of carbon, nitrogen, phosphorus, sulfur, iron, manganese and trace elements in wetlands. Chapter 3 describes various types of wetland vegetation and provides a brief description of plant adaptations to waterlogged conditions and growth parameters of macrophytes. The fourth chapter provides information about various types of constructed wetlands used for wastewater treatment. For each type, brief descriptions of major design parameters, together with application examples, are presented. The fifth chapter focuses on horizontal sub-surface flow constructed wetlands. Major design parameters such as pretreatment, water distribution and collection, filtration materials, vegetation, sizing and costs are described. Special attention is paid to the evaluation of treatment performance of HF CWs with respect to major pollutants in various types of wastewater. Chapter 6 provides information on the use of HF CWs for various types of wastewater.

The final chapter reviews the use HF CWs around the world. Information from 56 countries is included. The volume of scientific literature on constructed wetlands has grown immensely in recent years and our survey revealed that more than 100 international journals have published papers on constructed wetlands. Obviously, while it is not possible to gather all the information into one book, and the book cannot bring the complete information about the use of constructed wetlands in every country, the representative sampling will provide a thorough picture of the science around the world. Also, there is considerably more information on HF CWs in some countries. However, we tried to balance the length of the material on individual countries. For more detailed information readers can use sources listed in the *Suggested Reading* section at the end of the volume.

We would like to thank many colleagues who kindly provided their photos, which definitely enrich the content of the book. We also appreciate the help of our colleagues who either kindly provided unpublished materials or were helpful in gathering materials from their own countries, namely Marco Belmont, Suresh Billore, Jacques Brisson, Hans Brix, Tjaša Bulc, David Cooper, Paul Cooper, Verissimo Dias, Nathalie Fonder, Magdalena Gajewska, Joan Garcia, Roberta Gorra, Raimund Haberl, Tom Headley, Peter Horvát, Petr Hrnčíř, Frank Kansiime, Kunihiko Kato, Els Lesage, Ülo Mander, Fabio Masi, Jaime Nivala, John Pries, Gabrielle Mitterer-Reichmann, Silvana Perdomo, Diederik Rousseau, Chris Tanner, Karin Tonderski, Frank van Dien, Gladys Vidal, Scott Wallace and Róbert Zvara. We would also like to sincerely thank Paul Cooper, who carefully reviewed the manuscript and also corrected the language of the book, and Betty van Herk from Springer, for her excellent editorial cooperation.

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November 2007

Jan Vymazal, Praha, Czech Republic

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Chapter 1

INTRODUCTION

1. 1 Wetland values and functions

Wetlands have played a crucial role in human history. Major stages of the evolution of the life itself probably took place in nutrient-rich coastal waters. Some of the first prehistoric cultures, such as those of the early mesolithic settlements around the post-glacial lake margins and coasts of Europe and those of the coastal Indian communities in North America, depended on wetlands for food and materials for building, shelter and clothing (Maltby, 1991). Boulé (1994) in his excellent overview on an early history of wetland ecology pointed out that the early Sumerians knew the names of plants and animals that occupied the marshes of the Tigris and Euphrates Rivers, as evidenced by clay tablets on which those names were inscribed (Kramer, 1981). The Babylonians, who followed the Sumerians in Mezopotamia, not only had names for wetland plant species, but also established municipal reed beds and reeds harvested from these beds were used to make rugs, coarse mats to strengthen walls of clay brick, and very fine mats to serve as a foundation for dikes made from material dredged from the rivers (the original filter fabric).

Understanding regional hydrology was crucial to the success of both Mesopotamian, and later Egyptian, cultures. Not only did this knowledge make extensive agricultural enterprises possible, it also allowed for the creation of water gardens in the homes of the wealthy and powerful (Boulé, 1994). Early agricultural and horticultural experiments that led to the cultivation of wetland plant species are a major part of the origins of wetland science. Rice cultivation in China originated about 5000 B.C., while the oldest paddies have been dated at about 800 B.C. (Needham et al., 1986). By 1300 B.C. the municipal reeds were established in Babylon. Some evidence also exists for the ancient introduction of papyrus to Italy, although once cultivation ceased, it was gradually extirpated (Pickering, 1879).

The value of a wetland is a measure of its importance to society. Wetland functions are valued to various degrees by society, but there is no precise, general relationship between wetland functions and the value of wetlands to society, and values can be difficult to determine objectively. A wetland's value can be weighed directly or relative to other uses that could be made of the site; thus, the location of a wetland affects its value to society (Lewis 1995).

Wetlands are transitional environments. In a spatial context, they lie between dry land and open water – at the coast, around inland lakes and rivers, or as mires draped across the landscape. In an ecological context, wetlands are intermediate between terrestrial and aquatic ecosystems. In a temporal context, most wetlands are destined either to evolve into dry land as a result of lowered water tables, sedimentation and plant succession, or to be submerged by rising water tables associated with relative sea-level rise or climatic change. Wetlands often form part of a large continuum of community type, and therefore it is difficult to set boundaries. Consequently, few definitions adequately describe wetlands with the problem of definition usually arising on the edges of wetland, toward either wetter or drier conditions (Vymazal, 1995a).

As wetlands were considered as neither "true" terrestrial ecosystem nor "true" aquatic ecosystems, not many researchers were interested in wetland ecosystems. However, there has been an explosive growth of knowledge about, and a radical change of attitude toward wetlands since the 1950s (Williams, 1990). Wetlands have been recognized as providing many benefits including water supply and control (recharge of groundwater aquifers, drinking water, irrigation, flood control, water quality and wastewater treatment), mining (peat, sand, gravel), use of plants (staple food plants, grazing land, timber, paper production, roofing, agriculture, horticulture, fertilizers, fodder), wildlife (e.g. breeding grounds for waterfowl, preservation of flora and fauna), fish and invertebrates (shrimps, crabs, oysters, clams, mussels), integrated systems and aquaculture (e.g. fish cultivation combined with rice production), erosion control, gene pools and diversity, energy (hydroelectric, solar energy, heat pumps, gas, solid and liquid fuel), education and training, recreation and reclamation (Maltby, 1986; Löffler, 1990; Sather et al., 1990; Larson, 1990; Whigham and Brinson, 1990; Tinner, 1999; Mitsch and Gosselink, 2000).

Wetlands are not easily defined because they have a considerable range of hydrologic conditions, because they are found along a gradient at the margins of well-defined uplands and deepwater systems, and because of their great variation in size, location, and human influences. Wetland definitions, then, often include three major components (Mitsch and Gosselink, 2000):

1.Wetlands are distinguished by the presence of water, either at the surface or within the root zone.

2. Wetlands often have unique soil conditions (hydric soils) that differ from adjacent uplands.

3. Wetlands support vegetation adapted to the wet conditions (hydrophytes) and, conversely, are characterized by an absence of flooding-intolerant vegetation.

However, Mitsch and Gosselink (2000) pointed out that although the concepts of shallow water or saturated conditions, unique wetland soils, and vegetation adapted to wet conditions are fairly straightforward, combining these three factors to obtain a precise definition is difficult because of number of characteristics that distinguish wetlands from other ecosystems yet make them less easy to define.

1.2 Natural and constructed wetlands for wastewater treatment

Natural wetlands are characterized by extreme variability in functional components, making it virtually impossible to predict responses to wastewater application and to translate results from one geographical area to another. Although significant improvement in the quality of the wastewater is generally observed as a result of flow through natural wetlands, the extent of their treatment capability is largely unknown (Brix, 1993a). While most of natural wetland systems were not designed for wastewater treatment, studies have led to both a greater understanding of the potential of natural wetland ecosystems for pollutant assimilation and the design of new natural water treatment systems (Pries, 1994). It has only been during the past few decades that the planned use of wetlands for meeting wastewater treatment and water quality objectives has been seriously studied and implemented in a controlled manner. The functional role of wetlands in improving water quality has been a compelling argument for the preservation of natural wetlands and the construction of wetland systems for wastewater treatment (Bastian, 1993). Constructed wetlands can be built with a much greater degree of control, thus allowing the establishment of experimental treatment facilities with a well-defined composition of substrate, type of vegetation, and flow pattern. In addition, constructed wetlands offer several additional advantages compared to natural wetlands, include site selection, flexibility in sizing, and most importantly, control over the hydraulic pathways and retention time. The pollutants in such systems are removed through a combination of physical, chemical, and biological processes including sedimentation, precipitation, adsorption to soil particles, assimilation by the plant tissue, and microbial transformations (Brix, 1993a).

Natural wetlands have been used for wastewater treatment for centuries. In many cases, however, the reasoning behind this use was disposal, rather than treatment and the wetland simply served as a convenient recipient that was closer than the nearest river or other waterway (Wentz, 1987). Uncontrolled discharge of wastewater led in many cases to an irreversible degradation of many wetland areas. Wetlands have been considered for a long time as “wastelands”, were scientifically neglected and, therefore, the impact of wastewaters on different wetlands was not properly assessed. Cooper and Boon (1987), for example, pointed out that the use of natural wetlands for treatment of wastewater has been practiced in the United Kingdom for more than a century. In 1877, it was reported (Stanbridge, 1976) that a 6 m^3 of sewage was being applied daily per m^2 of land resulting in the production of an offensive-smelling swamp which produced a highly-polluted effluent. By providing suitable under-drainage, at a depth of about 1.8 m, it was possible to treat effectively about 50 liters of sewage daily per m^2 of land without the soil becoming clogged (Stanbridge, 1976).

Natural wetlands are still used for wastewater treatment uder controlled conditions (e.g. Kadlec and Tilton, 1979; Chan et al., 1982; Ewel et al., 1982; Olson, 1993, Mander and Jenssen, 2002) but the use of constructed wetlands has become more popular and effective around the world since the 1980s (e.g. Reddy and Smith, 1987; Hammer, 1989a; Cooper and Findlater, 1990; Moshiri, 1993; Kadlec and Knight, 1996; Vymazal et al., 1998; Kadlec et al., 2000; Mander and Jenssen, 2003).

Humans depend upon a symbiotic relationship between green plants and microorganisms for existence on earth. Photosynthetizing plants produce oxygen and regulate its atmospheric concentration while transforming radiant energy into useful chemical energy. In the process, carbon dioxide and other gaseous chemicals produced by humans, animals, and microorganisms during their metabolic processes are used and their atmospheric concentrations mediated. Plants in conjunction with microorganisms therefore produce food for humans and also recycle their wastes. These fundamental facts have been known for a long time and taken for granted. What has not been known for a long time is the potential of plants in conjunction with microorganisms for correcting environmental imbalances caused by industrial development and environmental abuse (Wolverton, 1987).

Constructed wetland treatment systems are engineered systems that have been designed and constructed to utilize the natural processes involving wetland vegetation, soils, and their associated microbial assemblages to assist in treating wastewater. They are designed to take an advantage of many of the same processes that occur in natural wetlands, but do so within a more controlled environment. Constructed wetlands consist of former terrestrial environment that have been modified to create poorly drained soils and wetlands flora and fauna for the primary purpose of contaminant or pollution removal from wastewater. Constructed wetlands are essentially wastewater treatment systems and are designed and operated as such, though many systems do support other functional values. Synonymous terms to constructed include man-made, engineered, and artificial wetlands (Hammer and Bastian, 1989a).

Wolverton (1987) pointed out that the scientific basis for wastewater treatment in a vascular aquatic plant system is the cooperative growth of both the plants and the microorganisms associated with the plants. A major part of the treatment process for degradation of organics is attributed to the microorganisms living on and around the plant root systems. Once microorganisms are established on aquatic plant roots, they form a symbiotic relationship in most cases with the higher plants. This relationship normally produces a synergistic effect resulting in increased degradation rates and removal of organic compounds from the wastewater surrounding the plant root systems. Also, microorganisms can use some or all metabolites released through plant roots as a food source. By each using the others waste products, this allows a reaction to be sustained in favor of rapid removal of organics from wastewater.

The first experiments aimed at the possibility of wastewater treatment by wetland plants were undertaken by Käthe Seidel in Germany in 1952 at the Max Planck Institute in Plön (Seidel, 1955). From 1955, Seidel carried out numerous experiments on the use of wetland plants, and especially Bulrush (*Schoenoplectus* = *Scirpus lacustris*), for treatment of various types of wastewater (Seidel 1961, 1965a, 1966, 1976). Although Seidel's experiments were heavily criticized (e.g. Nümann, 1970), many researchers continued in her ideas. The major reason for the criticism was the fact that investigations and calculations were mostly aimed only at the use of plants for nutrient removal by plant uptake. This would have required a regular harvest regime (which is not easy in many cases) and large areas needed for aquatic plants growth.

Wissing (1995) mentioned that with the application of the sewage treatment by activated sludge during the 1950s and 1960s on a large scale in German cities, Seidel recognized the arising problem of mounting contaminated sewage sludges from centralized treatment plants. She intensified her trial to grow helophytes and hydrophytes in wastewater and sludge of different origin and she tried to improve the performance of rural and decentralized wastewater treatment facilities which were either septic tanks or pond systems with poor cleaning effect. She planted macrophytes into the shallow embankment of tray-like ditches and created artificial trays and ditches grown with macropyhtes. Seidel names this early system the “hydrobotanical method”.

However, at that time, views on wastewater treatment among experts were limited to physical, chemical and biological (bacterial) methods and the controlled use of macrophytes for water purification was not taken into consideration. In addition, it was believed that most macrophytes cannot grow well in polluted water and the ability of macrophytes to eliminate toxic substances in water was not recognized as well (Seidel, 1976). Seidel’s concept to apply macrophytes to sewage treatment was difficult to understand for sewage engineers who had eradicated any visible plants on a treatment site for more than 50 years (Börner et al., 1998) and therefore, it was no surprise that the first full-scale constructed wetlands were built outside Germany.

In spite of many prejudices among civil engineers about odour nuisance, attraction of flies, poor performance in cold periods the IJssel Lake Polder Authority in Flevoland in The Netherlands constructed its first free water surface constructed wetland (FWS CW) in 1967 (de Jong, 1976; Greiner and de Jong, 1984; Veenstra, 1998). In 1968, FWS CW was created in Hungary near Keszthely in order to preserve the water quality of Lake Balaton and to treat wastewater of the town (Lakatos, 1998). However, FWS CWs did not spread significantly throughout the Europe as much as in North America (see section 4.1.4). Instead, constructed wetlands with sub-surface flow drew more attention in Europe; during the 1980s CW with horizontal flow (for details see Chapters 5, 6 and 7) and in 1990s also with vertical flow (see section 4.2.2) and their combinations (Cooper et al., 1996; Vymazal et al., 1998; Vymazal, 1999b, 2001a, 2003a, 2005c; see also section 4.3).

In North America, the free water surface wetland technology started with the ecological engineering of natural wetlands for wastewater treatment. Between 1967 and 1972, Howard T. Odum of the University of North Carolina, Chapel Hill, began a study using coastal lagoons for recycling and reuse of municipal wastewaters (Odum, 1985). In 1972, Odum, who had relocated to the University of Florida in Gainesville, began with Katherine Ewel to study the effectiveness of natural cypress wetlands for municipal wastewater recycling (Odum et al., 1977, Ewel and Odum, 1984). About at the same time, researchers at the University of Michigan in Ann Arbor began the Houghton Lake project, the first in-depth study using engineered wetlands for wastewater treatment in a cold climate region (Kadlec et al., 1975, Kadlec and Tilton, 1979). Since then constructed wetlands with free water surface have been used in North America for various types of wastewater including municipal sewage and industrial and agricultural effluents (e.g., Kadlec and Knight, 1996; Kadlec, 2003).

The sub-surface technology was started in North America during the early 1970s. (Spangler et al., 1976; Fetter et al., 1976; Small and Wurm, 1977, see section 7.2.3). In recent years the use of these systems has drawn more attention and it is estimated that there are about 8 000 subsurface constructed wetlands at present (Kadlec, 2003). However, the information on these systems is quite sparse as compared to free water CWs.

During the late 1970s and early 1980s, there was an explosion of research studies on the use of Water hyacinth (*Eichhornia crassipes*) for wastewater treatment (e.g., Bastian and Reed, 1979; Reed and Bastian, 1980; Reddy and Smith, 1987; Reed et al., 1988). However, after this period the interest disappeared because these systems proved to be difficult to manage and very costly in operation.

The potential use of aquatic and wetland macrophytes for wastewater treatment was evaluated in Australia by Mitchell during the mid 1970s (Mitchell, 1976). In 1980, the assimilative capacity of wetlands for sewage effluent was evaluated (Bavor et al., 1981) and Finlayson and co-workers performed pilot-scale experiments on the use of sub-surface constructed wetlands for the treatment of piggery wastes and abattoir wastewater (Finlayson and Chick, 1983; Finlayson et al., 1987). Extensive pilot-scale experiments were also carried out at University of Western Sydney (Bavor et al., 1987). At present, constructed wetlands in Australia are predominantly used for stormwater runoff treatment (free water surface CWs) but other applications could also been found including sub-surface systems.

Tanner et al. (2000) reported that constructed wetlands had been adopted enthusiastically by many New Zealand communities as a cost-effective means of secondary and tertiary wastewater treatment. The survey revealed that there were mere than 80 constructed wetlands for wastewater treatment excluding those treating stormwaters and farm dairy wastes. Surface flow CWs were most common (45%) followed by subsurface flow and hybrid systems (35% and 14%, respectively). At present, constructed wetlands in New Zealand are also very often used to treat agricultural runoff waters.

Since the mid 1980s, the concept of using constructed wetlands has gained increasing support in Southern Africa. By 1990, there were approximately 30 systems either in operation or under construction. These have been designed to serve a number of functions from treating raw sewage and secondary domestic effluents, upgrading septic tank and oxidation pond effluents, storm waters, agricultural and aquaculture wastes and a variety of industrial and mining wastewaters. Several of the systems have been constructed on the "root-zone" principles, other systems incorporated surface or vertical flow (Wood, 1990; Wood and Hensman, 1989). However, after the mid 1990s, the information from the South Africa diminished so it is not possible to find out if constructed wetlands became more widely spread there. On the other hand, at the end of the 20th century constructed wetland became more popular in tropical parts of Africa and there are now many fine examples of all types of constructed wetlands treating municipal sewage as well as industrial wastewaters and mine drainage waters in (Proceedings, 1998, 2000, 2002, 2004).

The traditional expertise of Asian farmers in recycling human and animal wastes through aquaculture and the practices intuitively developed by them for recovering nutrients from wastes by aquatic macrophytes propagated over waste-fed ponds gave a good basis for more engineered systems (Abassi, 1987). As early as in 1969, Sinha and Sinha reported on the use water hyacinth to treat digested sugar factory wastes. During the 1970s and 1980s numerous experiments with Water hyacinth were conducted across Asia to treat various types of wastewater, e.g. from dairies, palm oil production, distillery, natural rubber production, tannery, textile, electroplating, pulp and paper production, pesticide production and heavy metals (Abassi, 1987). However, the first information about the use of constructed wetlands with emergent vegetation appeared only in the early 1990s (Juwarkar et al., 1992). During the IWA conference in China in 1994, many papers on both horizontal and vertical flow CWs from Asia, and especially China, were presented and, therefore, it is probably a lack of literature information which made the Asian systems “unrecognized”. At present, CWs are in operation, among others, in India, China, Korea, Taiwan, Japan, Nepal, Malaysia or Thailand (Proceedings, 1998, 2000, 2002, 2004) for various types of wastewater.

Since 1980, research has been conducted in Brazil on the possibility of the use of water hyacinth ponds in combination with constructed wetlands planted with rice, here called “filtering soil” (Salati, 1987). Under current classification, these systems would be called vertical upflow CWs. However, other types of constructed wetlands with emergent macrophytes have been adopted recently (Proceedings, 2000). The information on the use of constructed wetlands with emergent vegetation in South America is limited but these systems are apparently in operation Brazil, Colombia, Ecuador, Uruguay, Argentina (e.g. Proceedings, 1998; Hadad et al., 2006; Perdomo, pers. comm.) and also in Central America (e.g., Platzer et al., 2002).

The very early attempts to use wetland macrophytes for water treatment were aimed at removal of various chemical compounds (Table 1-1). However, over the years constructed wetlands have primarily been used to treat municipal or domestic wastewaters. At present, constructed wetlands are used to treat all kinds of wastewaters including those from industrial and agricultural operations, stormwater runoff or landfill leachates (Table 1-1).

The first European national guideline was published in Germany by ATV (Abwassertechnische Vereinigung) in 1989 (ATV H 262, 1989) followed by European Guidelines (Cooper, 1990). At present, there are some kind of

Table 1-1. Examples of the first use of macrophytes and/or constructed wetlands for the treatment of different types of pollution (EXP = experimental, OP = operational). Updated from Vymazal et al. (1998a), with permission from Backhuys Publishers.

1952 - phenol wastewaters - EXP (Seidel, 1955, 1965a, 1966)
1956 - dairy wastewater - EXP (Seidel, 1976)
1956 - livestock wastewater - EXP (Seidel, 1961)
1965 - sludge dewatering - EXP (Bittmann and Seidel, 1967)
1967 - sewage OP (De Jong, 1976)
1973 - textile wastewater - EXP (Widyanto, 1975)
1974 - sludge dewatering - OP (Neurohr, 1983)
1975 - oil refinery wastewaters - OP (Litchfield and Schatz, 1989)
1975 - photographic laboratory wastewaters - EXP (Wolverton and McDonald, 1976)
1978 - textile mill wastewaters - OP (Kickuth, 1982a)
1978 - acid mine drainage - EXP (Huntsman et al., 1978)
1979 - fish rearing pond discharge - OP (Hammer and Rogers, 1980)
1980 - electroplating wastewater - EXP (Shroff, 1982)
1980 - removal of cresol - EXP (Wolverton and McDonald, 1981)
1980 - piggery effluent - EXP (Finlayson et al., 1987)
1980 - abattoir wastewater - EXP (Finlayson and Chick, 1983)
1981 - heavy metals removal - EXP (Gersberg et al., 1984)
1981 - tannery wastewater - EXP (Prasad et al., 1983)
1982 - acid mine drainage - OP (Stone, 1984; Pesavento, 1984)
1982 - agricultural drainage effluents - EXP (Reddy et al., 1982)
1982 - urban stormwater runoff - OP (Silverman, 1989)
1982 - pesticides - EXP (Gudekar et al., 1984)
1982 - sugar refinery wastewater - EXP (Yeoh, 1983)
1982 - benzene and its derivatives - EXP (Wolverton et al., 1984a)
1982 - rubber industry effluent - EXP (John, 1984)
1983 - rubber industry effluent - OP (John, 1984)
1983 - pulp/paper mill wastewaters - EXP (Allender, 1984; Thut, 1989, 1990a)
1985 - dairy wastewaters - OP (Brix and Schierup, 1989a)
1985 - seafood processing wastewater - EXP (Guida and Kugelman, 1989)
1986 - potato starch industry wastewater - EXP (De Zeeuw et al., 1990)
1986 - seepage from piled pig muck - OP (Gray et al., 1990)
1986 - cyanides and chlorphenols - EXP (Wolverton and Bounds, 1988)
1986 - ash pond seepage - OP (Brodie et al., 1989)
1987 - thermally affected wastewater - OP (Ailstock, 1989)
1987 - meat processing effluent - EXP (van Oostrom and Cooper, 1990)
1988 - landfill leachate - EXP (Staubitz et al., 1989; Birkbeck et al., 1990)
1988 - livestock wastewaters - OP (Hammer, 1989b; Hammer, 1992)
1988 - pulp/paper mill wastewater - OP (Thut, 1990b, 1993)
1989 - landfill leachate - OP (Surface et al., 1993)
1989 - agricultural runoff - OP (Higgins et al., 1993)
1989 - reduction of lake eutrophication - OP (Szilagyi et al., 1990)
1989 - chicken manure - EXP (Vymazal, 1990)
1990 - water from a swimming area in the lake OP (Vincent, 1992)
1991 - fish aquaculture EXP (Zachritz and Jacquez, 1993)
1991 - phenanthrene EXP (Machate et al., 1997)
1991 - woodwaste leachate - OP (Hunter et al., 1993)
1992 - bakery wastewater - OP (Vymazal, 1994)
1992 - sugar beet processing wastewaters - OP (Anderson, 1993)
1992 - combined sewer overflow OP (Cooper et al., 1996)

1993 - pesticides contaminated agricultural runoff OP (Braskerud and Haarstad, 2003)
1993 - highway runoff - OP (Swift a Landsdown, 1994)
1994 - abattoir wastewaters - OP (Vymazal, 1998)
1994 - glycol contaminated airport runoff - OP (Worrall, 1995)
1994 - poultry wastewaters OP (Hill and Rogers, 1997)
1994 - hydrocarbons EXP (Salmon et al., 1998)
1994 - urban surface water outfalls - OP (Scholes et al., 1995)
1995 - lignite pyrolysis wastewater EXP (Wiessner et al., 1999)
1995 - greenhouse wastewaters OP (Prystay and Lo, 1996)
1995 - nitroaromatic organic compounds OP (Novais and Martins-Dias, 2003)
1996 - explosives OP (Best et al., 2000; Behrends et al., 2000)
1997 - hydrocarbons (TPH/BTEX) OP (Moore et al., 2000a)
1998 - trout farm effluent OP (Comeau et al., 2001)
1998 - coke plant effluent EXP (Jardinier et al., 2001)
1998 - golf course runoff OP (Kohler et al., 2004)
1998 - nylon intermediates and ethylene based polymers OP (Snyder and Mokry, 2000)
1999 - molasses based distillery effluent OP (Billore et al., 2001)
2000 - winery wastewater OP (Masi et al., 2002; Rochard et al., 2002)
2000 - linear alkylbenzensulfonates (LAS) EXP (Del Bubba et al., 2000)
2000 - steel processing industry wastewaters EXP (Yang et al., 2002)
2000 - subsurface drainage from grazed dairy pastures OP (Tanner et al., 2003)
2001 - brewery wastewater EXP (Kalibbala et al., 2002)
2002 - tool factory wastewaters - OP (Maine et al., 2006)
2003 - olive mill wastewater OP (Kapellakis et al., 2004)
2003 - azo dyes EXP (Davies et al., 2005)
2003 - Endocrine Disrupting Chemicals OP (Masi et al., 2004)*
2004 - chlorobenzene EXP (Braeckevelt et al., 2006)

*not especially built for this purpose

guidelines for design and operation of constructed wetlands in most European countries. In some countries, such as Denmark, the guidelines have been issued for various types of constructed wetlands (horizontal flow, vertical flow, willow systems) separately (Ministry of Environment and Energy, 1999, 2003a, 2003b; Brix and Johansen, 2004). In the United States, a design manual on constructed wetlands and aquatic plant systems for municipal wastewater treatment was issued by U.S. Environmental Protection Agency in 1988 (U.S. EPA, 1988). The manual was replaced with an updated manual in 2000 (U.S. EPA, 2000). In Australia, Guidelines for using FWS constructed wetlands for the municipal sewage treatment were issued in 2000 (QDNR, 2000).

Chapter 2

TRANSFORMATION MECHANISMS OF MAJOR NUTRIENTS AND METALS IN WETLANDS

The three most important physicochemical properties of the soil that are affected by flooding are pH value, ionic strength, and oxidation-reduction potential (Eh or redox potential) (Patrick et al., 1985).

Wetland soils and overlying waters occur in a wide range of pH values. Organic soils in wetlands are often acidic, particularly in peatlands in which there is little groundwater inflow. On the other hand, mineral soils often have more neutral or alkaline conditions (Mitsch and Gosselink, 2000). The pH of most soils tend to change toward the neutral point after flooding, with acidic soils increasing and alkaline soils decreasing in pH. Increases as great as 3 pH units have been measured in some acid soils. The equilibrium pH for waterlogged soils is usually between pH 6.5 and 7.5 (Patrick et al., 1985). The tendency of soils of low pH to decrease in acidity and for soils of high pH to increase in acidity when submerged indicates that the pH of a submerged soil is buffered around neutrality by substances produced as a result of reduction reactions. Among the more likely compounds involved in buffering the pH of waterlogged soils are Fe and Mn compounds in the form of hydroxides and carbonates, and carbonic acid (Patrick et al., 1985). For some organic soils high in iron content, submergence does not always increase pH (Ponnamperuma, 1972). Peat soils often remain acidic during submergence through the slow oxidation of sulfur compounds near the surface, producing sulfuric acid and the production of humic acids and selective cation exchange by *Sphagnum* moss (Mitsch and Gosselink. 2000).

Flooding the soil causes an increase in the concentration of ions in the soil solution, although the increase may not persist throughout the growing season. In slightly acid and acid soils, the reduction of insoluble Fe, and possible Mn compounds, to more soluble forms accounts for much of the increase in cations. In neutral to slightly alkaline soils, Ca^{2+}, and Mg^{2+} in the soil solution make significant contributions to the ionic strength. Ferrous and manganous ions produced through reduction reactions displace other cations from the exchange complex to the soil solution (Patrick et al., 1985).

2.1 Oxygen and redox potential

In well drained soils, most of the pore spaces surrounding individual soil particles and aggregates are gas-filled and interconnected with the atmosphere. This permits relatively rapid gaseous diffusion of oxygen throughout the plant rooting depth. Though there may be a reduction in gaseous oxygen content with depth in some soils (Russell, 1961), there is sufficient molecular oxygen transport across the gas-liquid interface of the soil solution to maintain some dissolved oxygen in this solution. As a result, the soil is maintained in an oxidized condition. The potential oxygen re-supply rate by this process is usually more than sufficient to meet soil and root oxygen demand (Gambrell and Patrick, 1978).

Excess water applied to a permeable soil by precipitation, irrigation, or temporary flooding will rapidly drain from the upper profile through the interconnected pore spaces. Much of this pore space is again filled with gas, which is continuous with the atmosphere, after draining for several hours. When soils are inundated the pore spaces are filled with water and the rate at which oxygen can diffuse through the soil is drastically reduced. Diffusion of oxygen in an aqueous solution has been estimated at 10 000 times slower than oxygen diffusion through a porous medium such as drained soils (Greenwood, 1961; Greenwood and Goodman, 1964). As a result of prolonged flooding and continued oxygen demand for root and microbial respiration, as well as chemical oxidation of reduced organic and inorganic components, the oxygen content of the soil solution begins an immediate decline and may be depleted within several hours to a few days (Fig. 2-1). The rate at which the oxygen is depleted depends on the ambient temperature, the availability of organic substrates for microbial respiration, and sometimes the chemical oxygen demands from reductants such as ferrous iron. The resulting lack of oxygen prevents plants from carrying out normal aerobic root respiration and strongly affects the availability of plant nutrients in the soil. As a result, plants that grow in anaerobic wetland soils generally have a number of specific adaptations to these conditions (Mitsch and Gosselink, 2000).

It is not always true that oxygen is totally depleted from the soil water of wetlands. There is usually a thin layer of oxidized soil, sometimes only a few centimeters thick at the surface of the soil at the soil-water interface. The thickness of the oxidized layer is directly related to:

- the rate of oxygen transport across the atmosphere-surface water interface
- the small population of oxygen-consuming organisms present
- photosynthetic oxygen production by algae within the water column
- surface mixing by convection currents and wind action (Gambrell and Patrick, 1978).

The depth of the oxidized layer depends on a balance between the rate of oxygen diffusion into the surface horizon and its consumption (Mortimer, 1942). Oxygen consumption rates have been thought to be a function of microbial respiration. However, Howeler and Bouldin (1971) demonstrated that oxygen consumption rates in some flooded soils can best be described by models including oxygen consumption for both biological respiration and for chemical oxidation of both mobile and non-mobile constituents. Reduced

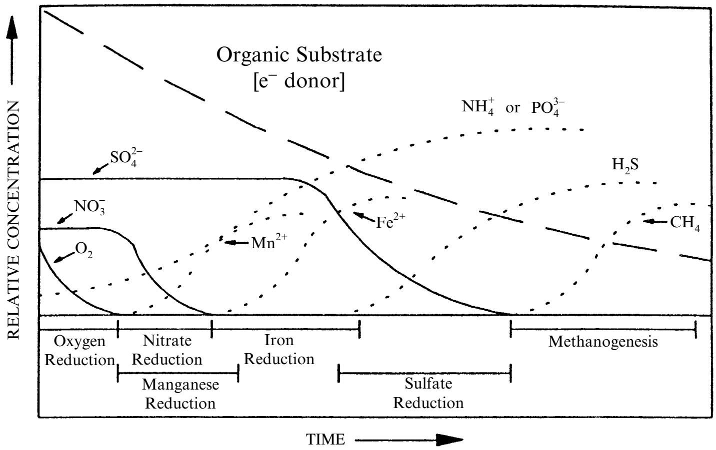


Figure 2-1. Sequence time of transformations in soil flooding, beginning with oxygen depletion and followed by nitrate and then sulfate reduction. Increases are seen in reduced manganese (manganous), reduced iron (ferrous), hydrogen sulfide, and methane. Note the gradual decrease in organic substrate (electron donor) and increase in available ammonium and phosphate ions. (After Reddy and D'Angelo, 1994, with permission from Elsevier).

iron and manganese ions were thought to represent the bulk of the mobile reductants while precipitated ferrous iron, manganous manganese and sulfide compounds, encountered as the oxidized zone increased in thickness, likely comprised much of the non-mobile constituents. Howeler (1972) pointed out that the ratio between biological and chemical oxygen consumption rates may vary widely depending on the organic matter content of the soil or sediment.

Oxygen diffusion is not the only route for oxygen transport in the flooded soil. It is well documented that aquatic and wetland macrophytes release oxygen from roots into the rhizosphere and that this release influences the biogeochemical cycles in the sediments through the effects on the redox status of the soils and sediments (e.g., Barko et al., 1991; Sorrell and Boon, 1992). Qualitatively, this is easily visualized by the reddish color associated with oxidized forms of iron on the surface of the roots. But the quantitative magnitude of the oxygen release under *in situ* conditions remains a matter of controversy (Bedford et al., 1991; Sorrell and Armstrong, 1994, Brix, 1998).

Oxygen release rates from roots depend on the internal oxygen concentration, the oxygen demand of the surrounding medium and the permeability of the root-walls (Sorrell and Armstrong, 1994). Wetland plants conserve internal oxygen because of suberized and lignified layers in the hypodermis and outer cortex (Armstrong and Armstrong, 1988). These stop radial leakage outward, allowing more oxygen to reach the apical meristem. Thus, wetland plants attempt to minimize their oxygen losses to the rhizosphere. Wetland plants do, however, leak oxygen from their roots (Brix, 1998). Rates of oxygen leakage are generally highest in the sub-apical region of roots and decrease with distance from the root-apex (Armstrong, 1979). The oxygen leakage at the root-tips serve to oxidize and and detoxify potentially harmful reducing substances in the rhizosphere. Species possessing an internal convective throughflow ventilation system have higher internal oxygen concentrations in the rhizomes and roots than species relying exclusively on diffusive transfer of oxygen (Armstrong and Armstrong, 1990), and the convective throughflow of gas significantly increases the root length by diffusion alone (Brix, 1994a). Wetland plants with a convective throughflow mechanism therefore have the potential to release more oxygen from their roots compared to species without convective throughflow.

Using different assumptions of root oxygen release rates, root dimensions, numbers, permeability, etc., Lawson (1985) calculated a possible oxygen flux from roots of *Phragmites australis* up to 4.3 g m^{-2} d^{-1}. Others, using different techniques, have estimated root oxygen release rates from *Phragmites* to be 0.02 g m^{-2} d^{-1} (Brix, 1990a; Brix and Schierup, 1990), 1-2 g m^{-2} d^{-1} (Gries et al., 1990) and 5-12 g m^{-2} d^{-1} (Armstrong et al., 1990). Root oxygen release rates from a number of submerged plants are reported to be in the range of 0.5 to 5.2 g m^{-2} d^{-1} (Sand-Jensen et al., 1982; Kemp and Murray, 1986; Caffrey and Kemp, 1991) and from free-floating plants 0.25 to 9.6 g m^{-2} d^{-1} (Moorhead and Reddy, 1988; Perdomo et al., 1996). Brix (1998) reported that gas exchange experiments in Denmark have shown that 4 g O_2 m^{-2} d^{-1} is transferred from the atmosphere to the soil. The reed vegetation transport 2 g O_2 m^{-2} d^{-1} to the root zone is mainly utilized by the roots and rhizomes themselves.

In the summer period, pressure may built up in the lacunar air spaces of the plants, which induces a mass flow of gasses internally in the plant and hence a better aeration of the buried root system (Brix et al., 1992). However, the roots and rhizomes also have a higher oxygen demand during summer because of the higher temperature. In natural *Phragmites* stands a net flux of up to 8 g O_2 m^{-2} d^{-1} through the reeds has been estimated (Brix et al., 1996). However, most of this oxygen is probably used to cover the respiratory demand of the root-rhizome system leaving only insignificant amounts of oxygen available for waste treatment processes (Brix, 1998).

Redox potential (Eh) is a measure of the electrochemical potential or electron availability in chemical and biological systems. Electrons are essential to all chemical reactions – chemical species that lose electrons become oxidized and conversely, reduction occurs as a chemical species gains electrons. Thus, a measure of redox potential (electron availability) indicates the intensity of oxidation or reduction of a chemical or biological system (Gambrell and Patrick, 1978). In an aqueous system, the intensity of oxidation is limited by the electrochemical potential at which water becomes unstable and releases molecular oxygen. Similarly, the potential at which molecular hydrogen is released from water represents the lower limit of reduction in aqueous systems (Baas Becking et al., 1960). Within the limits imposed by the stability of water, the oxidation states of hydrogen, carbon, nitrogen, oxygen, sulfur and several metals may be affected by the oxidation-reduction potential of a system, though the measured redox potential is dependent on the chemical activity of a few of the more abundant oxidized and reduced forms of these elements present (Bohn, 1971).

Non-photosynthetic biological activity in the soil derives energy from the oxidation or reduced substrates, which may be either organic (for heterotrophic metabolism) or inorganic (chemoautotrophic metabolism) in nature. Plants, of course, get this energy directly from sunlight (Killham, 1994). The metabolism of all living cells is an open system which is characterized by a continuous input and output of matter and energy. Each cell is endowed with a system that transforms the chemical and physical energy taken up into biological useful energy (ATP) and utilizes the latter to perform work (Thauer et al., 1977). It also should be noticed that energy utilization does not occur with 100% efficiency (Reddy et al., 1986). The oxidation of organic matter produced in photosynthesis yields energy; the amount of energy depends on the nature of oxidant, or electron acceptor. Energetically, the most favorable oxidant is oxygen; after oxygen is depleted there follows a succession of organisms capable of reducing NO_3^-, MnO_2, FeOOH, SO_4^{2-} and CO_2 with each oxidant yielding successively less energy for the organism mediating the reaction (Westall and Stumm, 1980).

Two important points pertaining to microbial activity in flooded soils vs. upland terrestrial soils that should be noted are that (Gambrell et al., 1991): 1) energy release from microbial utilization of soil organic matter is much more efficient under aerobic conditions than anaerobic conditions, and, 2) the organic and inorganic end products of microbial metabolic processes differ between aerobic and anaerobic respiration (Alexander, 1961; Reddy and Patrick, 1975; Tusneem and Patrick, 1971). Because of the lower energy efficiency, anaerobic organisms are less efficient in assimilating soil organic matter during decomposition, thus the rate of soil organic matter mineralization is less in soils with poor aeration (Acharya, 1935; DeLaune et al., 1981). This accounts for sediments, swamp soils and flooded field soils having greater organic matter content that upland soils in the same area (Gambrell et al., 1991).

Another important difference in microbial activity in anaerobic vs. aerobic soils is the end products of microbial metabolism. Anaerobic metabolism results in the formation of low molecular weight organic acids, complex residual humic materials, carbon dioxide, methane, hydrogen, ammonia, amines, mercaptanes, and hydrogen sulfide, though the formation of some of these depends on the intensity of reduction. Aerobic metabolism, on the other hand, results in mostly the formation of carbon dioxide, nitrate, sulfate, plus residual humic materials. It is believed the humic materials formed and transformed under anaerobic conditions may tend to have a larger molecular weight and be structurally more complex, factors that may affect the mobility of trace and toxic metals (Gambrell et al., 1980).

In addition to the difference in the end products of aerobic and anaerobic decomposition, there is a large disparity in the amount of energy released; this greater energy release allows a more efficient synthesis of cellular material per unit of organic nutrient. Under aerobic conditions, utilization of substrate C is relatively high, ranging from 20 to 40%, depending on the microbial population. Anaerobic bacteria typically realize a C assimilation rate of only 2 to 5%. Consequently, organic matter decomposition is retarded in flooded soils (Patrick et al., 1985).

Oxygen is the terminal electron acceptor in aerobic systems and is reduced while organic electron donors are being oxidized (Fig. 2-2A, Table 2-1). This reduction of O_2 to H_2O is carried out by true aerobic microorganisms and CO_2 is evolved as a waste product. Therefore, a supply of oxidizable organic compounds, as well as a supply of O_2 and some means of removing CO_2 produced are indispensable for aerobic respiration to occur (Reddy et al., 1986).

Aerobic soils have values of Eh between +300 and +800 mV and usually between +400 and +700 mV (Patrick and Mahapatra, 1968; Gambrell and Patrick, 1978; Reddy et al., 1986) and the CO_2 evolved during aerobic respiration diffuses relatively quickly when the air filled porosity is large. The relatively narrow range and poor reproducibility in oxidized systems

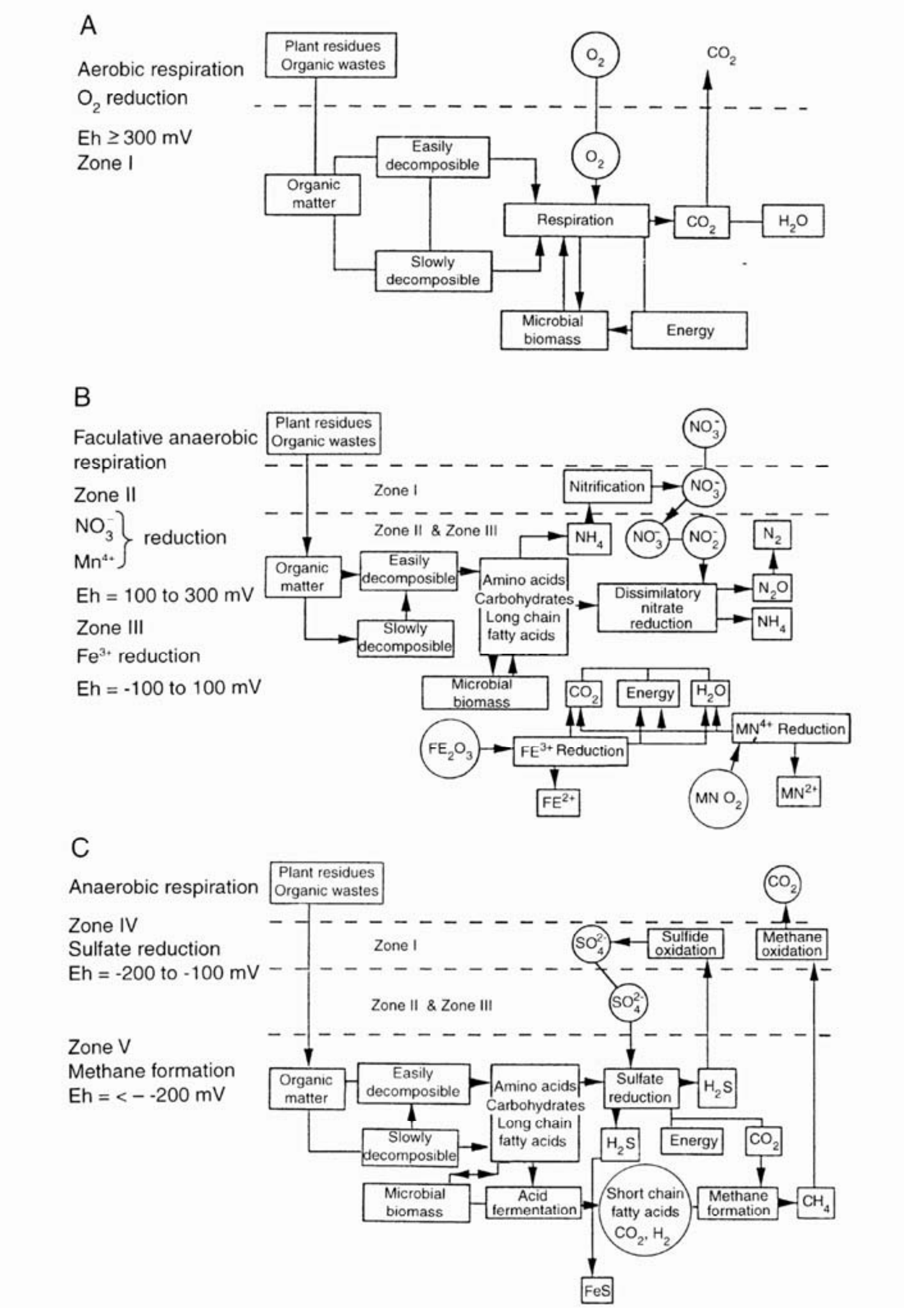


Figure 2.-2. Pathways of organic matter decomposition during aerobic respiration (A), facultative anaerobic respiration (B) and anaerobic respiration (C). From Reddy et al. (1986), with kind permission of Springer Science and Business Media.

limits the usefulness of redox potential measurements for characterizing aerobic soils and sediments. Oxygen content and oxygen diffusion rates are better indicators of oxidation intensity in these systems. In sediments and submersed soils, redox potential ranges from around -400 mV (strongly reduced) to +700 mV and is better poised and fairly reproducible at the more reduced levels. Thus, redox potential measurements have gained increasing acceptance as means of characterizing reduced soils and sediment-water systems (Gambrell and Patrick, 1978).

Table 2-1. Redox sequence of the biologically mediated oxidation of organic matter (Zehnder, 1980; Gambrell et al., 1991[a]; Killham, 1994; Mitsch and Gosselink, 2000[b]; Middelburg, 2000; Gambrell and Patrick, 1978[c], Laanbroek, 1990[d]).

Reaction	**Eh[a]**	**Eh[b]**	**Eh[c]**	**Eh[d]**
Aerobic respiration $CH_2O + O_2 \rightarrow CO_2 + H_2O$		+400 to +600	400 to +700	+330
Denitrification (nitrate respiration) $5CH_2O + 4NO_3^- \rightarrow 2N_2 + 4HCO_3^- + CO_2 + 3H_2O$	+225	+250	+220	+220
Manganese reduction $CH_2O + 3CO_2 + H_2O + 2MnO_2 \rightarrow 2Mn^{2+} + 4HCO_3^-$	+200	+225	+200	+200
Iron reduction $CH_2O + 7CO_2 + 4Fe(OH)_3 \rightarrow 4Fe^{2+} + 8HCO_3^- + 3H_2O$	+100	+100 to -100	+120	+120
Nitrate ammonification $CH_2O + 1/2NO_3^- + H^+ \rightarrow CO_2 + 1/2NH_4^+ + 1/2H_2O$	<-100			
Fermentation $3CH_2O \rightarrow CO_2 + C_2H_5OH$	-180			
Sulfate reduction (sulfate respiration) $CH_2O + SO_4^{2-} \rightarrow H_2S + 2HCO_3^-$	-150	-100 to -200	-150	-150
Methane formation $4H_2 + CO_2 \rightarrow CH_4 + 2H_2O$	-200	< -200	-250 to -300	-250

Zehnder (1980) points out that these values give only the redox potential for the overall reaction. Biological reactions, however, are mostly multi-step reactions. Each step may show a different equilibrium potential, compared to the one in the Table 2-1. Moreover the "milieu interieur" of the organisms differs quite markedly from the "milieu exterieur". Therefore, organisms may already carry out the listed reactions at much higher environmental redox potentials.

When the supply of O_2 to the soil is cut off, the obligate aerobic microorganisms can no longer function and so either die or go into a resting stage. The microbial community shifts to facultative anaerobic bacteria and when anoxic conditions persist obligate anaerobes also proliferate, while the fungal population decreases (Fig. 2-2B, Table 2-1). As O_2 is depleted, nitrate will be used as electron acceptor followed by oxidized manganese compounds and then followed by ferric iron compounds. The order of these reductions is the same as that indicated by thermodynamic considerations (Reddy et al., 1986).

The energy yield during NO_3^- reduction is slightly less than for O_2 as terminal electron acceptor (about 95% relative to O_2) and in order to release the same amount of energy 1.64 times the amount of O_2 would be needed using NO_3^-. Since the energy availability is quite large, it is not surprising to discover that there are a number of denitrifying bacteria capable of exploiting the use of NO_3^- as terminal electron acceptor (Reddy et al., 1986). Nitrate respiration was found to be about 1.7 times less efficient than O_2 respiration in studies with *Pseudomonas denitrificans* and *P. stutzeri* (Elliott and Gilmour, 1971; Koike and Hattori, 1975). Much lower energy yield (2.6 to 2.9 times less than O_2) was reported by Bryan (1980) for *P. aeruginosa* and *P. stutzeri*. Manganese reduction occurs at relatively high redox potentials (Table 2-1), and the energy yields about 67% relative to the O_2 and in order to release the same amount of energy 8.12 times the amount of O_2 would be needed using MnO_2 (Reddy et al., 1986). The Fe^{3+} compounds buffer the Eh of the system in the range of about -100 to +100 mV. The buffering effect of Fe is essentially depleted at about -150 mV (Reddy et al., 1986), and it is in this range that facultative anaerobes cease to function (Patrick, 1964). The energy yields about 15% relative to the O_2 and in order to release the same amount of energy about 91 times the amount of O_2 would be needed using $Fe(OH)_3$ (Reddy et al., 1986).

Anaerobic respiration is defined as the metabolic activity of true anaerobes which utilize organic matter as energy source, and SO_4^{2-} and CO_2 as electron acceptors (Fig. 2-2C, Table 2-1). Anaerobic respiration is predominant in soils which are under continuous flooding (Reddy et al., 1986). The SO_4^{2-} reduction occurs when the redox potential drops below -100 mV and the energy yields about 13% relative to the O_2. Sulfate has about the same O_2 equivalent as NO_3^- and if the energy yields are compared on ppm basis it can be noted that about 11 times more SO_4^{2-} is needed to release the same amount of energy in the system than when the organic matter is oxidized with O_2. In nature SO_4^{2-} reduction occurs in soils and sediments which have adequate amounts of easily decomposable organic matter and a supply of SO_4^{2-} ions and anoxic conditions free of O_2, NO_3^-, Mn^{4+}, and Fe^{3+} (Reddy et al., 1986).

Methane production requires extremely reduced conditions, with a redox potential below -200 mV, after other terminal electron acceptors have been reduced. Methanogenic bacteria, restricted to the archaebacteria, utilize H_2 as an electron source, but can also use formate (HCOH). Approximately 14% utilize acetic acid (CH_3COOH), and 28% use 1-C compounds such as methanol (CH_3OH), methylamine $(CH_3)_3N$ and dimethylsulfide $(CH_3)_2S$ (Paul and Clark, 1996).

2.2 Carbon transformations

Wetlands are major sinks for carbon, as such, they are typically characterized by accretion of organic matter. Net accumulation of organic carbon is based on the balance between primary production and heterotrophic respiration (Reddy and D'Angelo, 1997).

The major processes of carbon transformation under aerobic conditions are photosynthesis (Eq. 2.1) and respiration (Eq. 2.2) which dominate the aerobic horizons (aerial and aerobic water and soil) with H_2O as the major electron donor in photosynthesis and oxygen as the terminal electron acceptor in respiration:

$$6CO_2 + 12H_2O + \text{light} \rightarrow C_6H_{12}O_6 + 6O_2 + 6H_2O \qquad (2.1)$$

$$C_6H_{12}O_6 + 6O_2 \rightarrow 6CO_2 + 6H_2O + 12e^- + \text{energy} \qquad (2.2)$$

Primary production is an essential link in the soil carbon cycle as it is the main flux from the atmosphere to the soil biota and is defined as the rate at which energy is stored in photosynthetic and chemosynthetic activity of producer organisms in the form of organic substances. In general, in the soil system, it is the plants (particularly macrophytes) that contribute most to primary productivity. There are also, however, photoautotrophic blue-green algae (cyanobacteria) as well as the eukaryotic algae which can contribute to carbon dioxide fixation. Photoautotrophic and chemoautotrophic bacteria may also make a very minor contribution (Killham, 1994).

The bulk primary production becomes partitioned between herbivores and decomposers. The latter group consists of chemoorganotrophic microorganisms and invertebrate animals that utilize dead organisms and their residues (detritus) as carbon and energy sources. (Grant and Long, 1981,1985). On land only 5-10% of net primary production is consumed by herbivores, the rest being transformed by decomposers (Swift et al., 1979).

The degradation of organic matter by aerobic respiration (Eq. 2.2) is fairly efficient in terms of energy transfer. However, because of the anoxic nature of wetlands, anaerobic processes, less efficient in terms of energy transfer, occur in close proximity to aerobic processes. Two of the major anaerobic processes are fermentation and methanogenesis. Pathways of carbon during decomposition in wetlands are given in Figure 2-2. Typically, in a drained soil, oxygen can be used as an electron acceptor, during aerobic respiration. Upon flooding sequential reduction of electron acceptors occurs as a function of Eh (Table 2-1). Although redox reactions may be far from equilibrium, they tend to occur in the order of their energy yield. The oxidation of organic matter produced in photosynthesis yields energy; the amount of energy depends on the nature of oxidant, or electron acceptor. Energetically, the most favorable oxidant is oxygen with each oxidant yielding successively less energy for the organism mediating the reaction (Westall and Stumm, 1980).

Anaerobic respiration occurs in the soil zone below the Fe^{3+} reduction zone (Fig. 2-2). The process can be carried out in wetland soils by either facultative or obligate anaerobes. It represents one of the major ways in which high molecular weight carbohydrates are broken down to low molecular weight organic compounds, usually as dissolved organic carbon, which are, in turn, available to microbes (Valiela, 1984). Anaerobic degradation is a multi-step process. In the first step the primary end-products of fermentation are fatty acids such as acetic (Eq. 2.3), butyric, and lactic (Eq. 2.4) acids, alcohols (Eq. 2.5) and the gases CO_2 and H_2 (Reddy and Graetz, 1988; Vymazal, 1995a; Mitsch and Gosselink, 2000):

$$C_6H_{12}O_6 \rightarrow 3CH_3COOH \text{ (acetic acid)} + H_2 \tag{2.3}$$

$$C_6H_{12}O_6 \rightarrow 2CH_3CHOHCOOH \text{ (lactic acid)} \tag{2.4}$$

$$C_6H_{12}O_6 \rightarrow 2CO_2 + 2CH_3CH_2OH \text{ (ethanol)} \tag{2.5}$$

Acetic acid is the primary acid formed in most flooded soils and sediments. Strictly anaerobic sulfate-reducing bacteria (Eq. 2.6) and methane-forming bacteria (Eqs. 2.7 and 2.8) then utilize the end-products of fermentation and, in fact, depend on the complex community of fermentative bacteria to supply substrate for their metabolic activities. Both groups play an important role in organic matter decomposition and carbon cycling in wetland soil environments (Valiela, 1984; Grant and Long, 1985):

$$CH_3COOH + H_2SO_4 \rightarrow 2CO_2 + 2H_2O + H_2S \tag{2.6}$$

$$CH_3COOH + 4H_2 \rightarrow 2CH_4 + 2H_2O \tag{2.7}$$

$$4H_2 + CO_2 \rightarrow CH_4 + 2H_2O \tag{2.8}$$

Methane is an important gas evolving from flooded and saturated soils. The major pathways of methane formation are 1) decarboxylation of acetic acid at the expense of hydrogen (Eq. 2.7) and 2) reduction of electron acceptors (Eq. 2.8) or by a unique aceticlastic reaction (Grant and Long, 1985):

$$CH_3COOH \rightarrow CH_4 + CO_2 \tag{2.9}$$

Methane is either released to the atmosphere or is oxidized to CO_2 by methanotrophic bacteria as soon as it enters the oxic zone (Fig. 2-2). It has been found that in the submerged soils without plants 35% of the produced CH_4 was emitted to the atmosphere. The presence of plants stimulated methanogenesis in the submerged soil but also enhanced the CH_4 oxidation rates within the rhizosphere, so that only about 23% of the CH_4 produced was emitted (Holzapfel-Pschorn et al., 1986).

The emission of methane occurs due to ebullition and/or plant mediated transport (Dacey and Klug, 1979; Holzapfel-Pschorn et al., 1986; Stepniewski and Glinski, 1988). The contribution of the plant-mediated transport depends on the kind and development stage of the plant cover. It has been found that that in the unvegetated paddy field CH_4 was emitted almost exclusively by ebullition, whereas in the presence of well-established plant cover (rice, reed, weeds) 60-94% of CH_4 emission was due to plant-mediated transport (Stepniewski and Glinski, 1988).

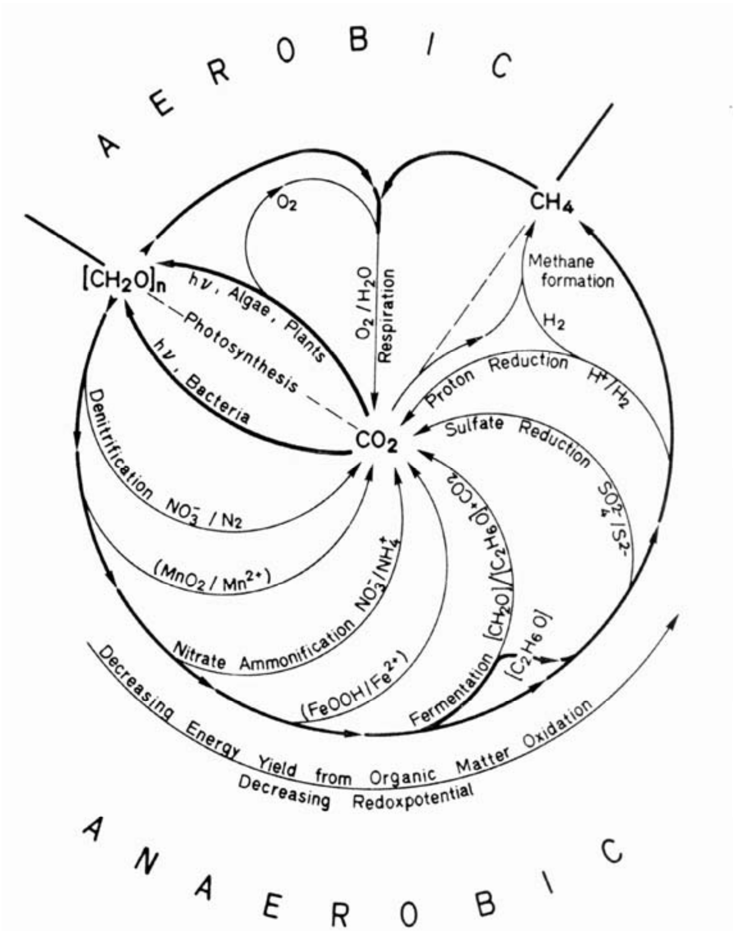


Figure 2-3. The biological CO_2 cycle. From Zehnder (1980), with kind permission of Springer Science and Business Media.

In general, methane is found at low concentrations in reduced soils if sulfate concentrations are high (Valiela, 1984). Possible reasons for this phenomenon include 1) competition for substrates that occurs between sulfur and methane bacteria, 2) the inhibitory effects of sulfate or sulfide on methane bacteria, or 3) a possible dependence of methane bacteria on products of sulfate-reducing bacteria (Gambrell and Patrick, 1978). From a biochemical perspective, methane is relatively difficult to activate. Its aerobic activation is catalyzed by monooxygenases, enzymes that make use of high-potential oxygen radical chemistry, unavailable to anaerobic life (Strous and Jetten, 2004). Furthermore, for a long time enrichment or detection of organisms capable of anaerobic growth on these compounds were unsuccessful (Sorokin, 1957). This led to the idea that methane was inert under anoxic conditions, a belief that persisted among many microbiologists throughout most of the twentieth century (Strous and Jetten, 2004). This has changed during the 1970s. Several independent studies showed that in marine sediments methane concentrations decreased from the sediments towards the water column (Barnes and Goldberg, 1976; Reeburgh, 1976; Martens and Berner, 1977; Zehnder and Brock, 1979). At the same time, sulfate concentrations decreased from the water column into the sediments, indicating that sulfate might be the electron acceptor for anaerobic methane oxidation (Eq. 2.10). During the early 1990s, it becomes clear that up to 90% of marine methane is oxidized anaerobically (Strous and Jetten, 2004; Megonikal et al., 2004).

$$CH_4 + SO_4^{2-} + H^+ = CO_2 + HS^- + 2H_2O \tag{2.10}$$

With the present conditions on the Earth organic matter is thermodynamically metastable. Thus, the ultimate fate of all carbon compounds is CO_2 (Fig. 2-3). The oxidation of organic compounds is catalyzed by heterotrophic organisms and the electrons are transferred to different acceptors depending on organism and environment; thereby chemical energy is released (see section 2.1).

2.3 Nitrogen transformations

Nitrogen has a complex biogeochemical cycle with multiple biotic and abiotic transformations involving seven valence states (+5 to -3). The compounds include a variety of inorganic and organic nitrogen forms that are essential for all biological life. Most of the soil N and sediments is in organic form with only a very small amount present in the inorganic form. A series of biochemical and physico-chemical processes are involved in transforming one source of N to another source. The most important forms of inorganic N compounds include ammonium (NH_4^+), nitrite (NO_2^-), nitrate

(NO_3^-), gaseous dinitrogen (N_2) and nitrous oxide (N_2O). These compounds are the end products of specific biological reactions. Nitrogen may also be present in wetlands in many organic forms including urea, amino acids,

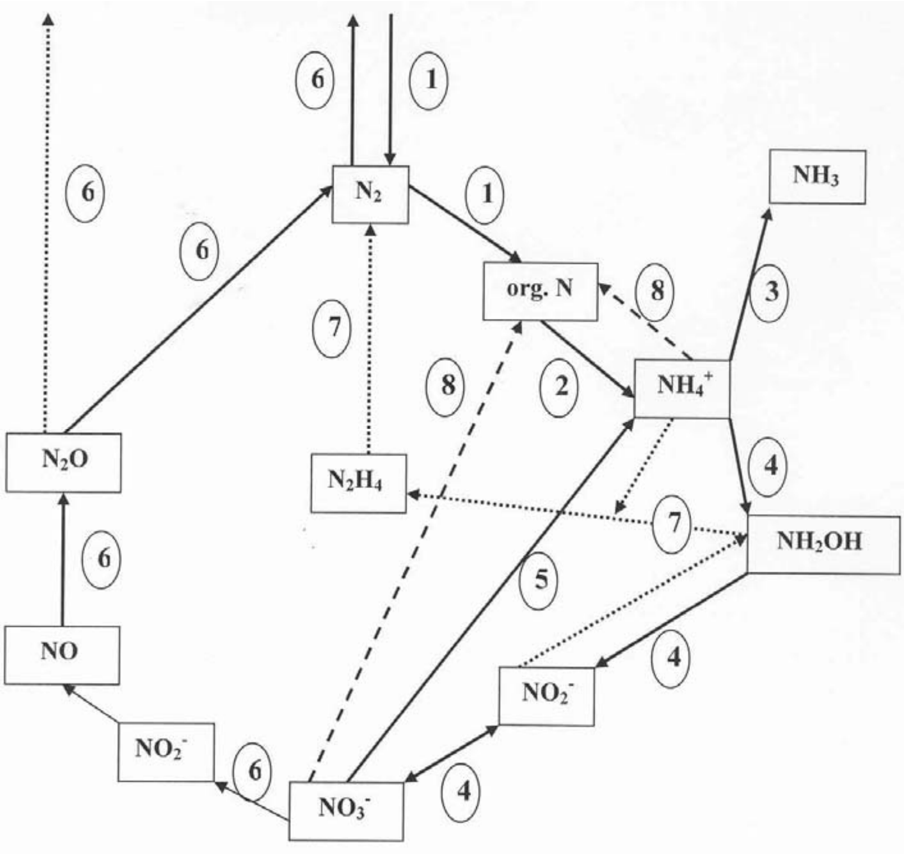


Figure 2-4. Major nitrogen transformations in wetlands. 1-fixation, 2-ammonification (mineralization), 3-ammonia volatilization, 4-nitrification, 5-nitrate-ammonification, 6-denitrification, 7-ANAMMOX, 8-uptake by plants and microbiota.

amines, purines, and pyrimidines (Vymazal, 1995). The most important conversion processes functioning in a wetland system are: ammonification (organic N → NH_4^+), nitrification ($NH_4^+ \rightarrow NO_2^- \rightarrow NO_3^-$) denitrification ($NO_3^- \rightarrow N_2O \rightarrow N_2$), biological fixation ($N_2$ → organic N), nitrate ammonification ($NO_3^- \rightarrow NH_4^+$), anaerobic ammonia oxidation (ANAMMOX, $NH_4^+ \rightarrow N_2$) and volatilization ($NH_4^+ \rightarrow NH_3$) (Figs. 2-4 and 2-5).

The microbial processes involved in nitrogen transformations have traditionally been described by aerobic nitrification, anoxic/anaerobic denitrification and aerobic/anaerobic ammonification. However, intensive studies, often aimed at processes responsible for nitrogen removal from wastewaters, revealed or confirmed many other microbial processes responsible for nitrogen transformations such as anaerobic ammonia oxidation, aerobic denitrification (Jetten et al., 1999; Jetten, 2001; Schmidt et al., 2003)

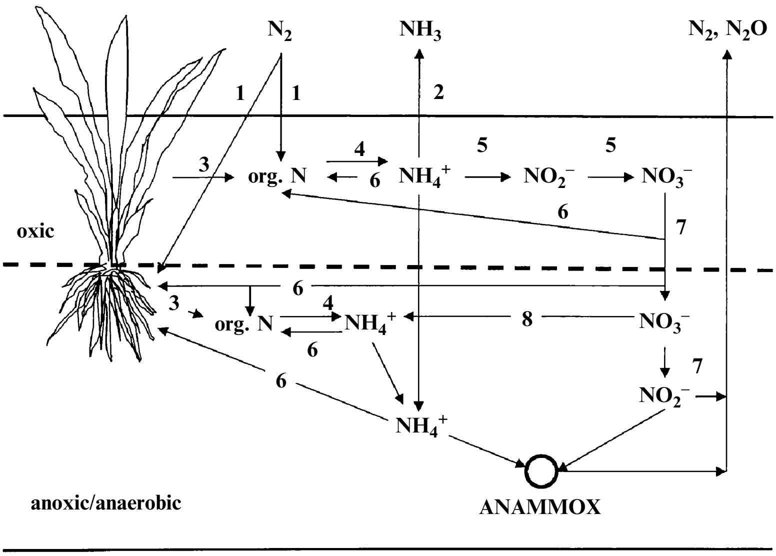


Figure 2-5. Major nitrogen transformations in aerobic (oxic) and anaerobic/anoxic zones of wetland systems. 1-N_2 fixation, 2-ammonia volatilization, 3-leaching, 4-ammonification (mineralization), 5-nitrification, 6-uptake, 7-nitrate diffusion and subsequent denitrification, 8-nitrate-ammonification,

2.3.1 Ammonification

Ammonification (mineralization) of organic nitrogen refers to the degradation of N-containing organic compounds such as proteins, amino sugars, and nucleic acids to NH_4^+, the mineral form. Whether the amino acids are used for energy source or as building block for proteins is dependent on a complex series of feedback controls. Carbohydrate C, if available, will usually be utilized, rather than the amino acid C, thus preserving the C skeleton of the amino acid for protein synthesis (Paul and Clark, 1996). A large fraction (up to 100%) of the organic nitrogen is readily converted to ammonia (Kadlec and Knight, 1996).

The ammonification process is essentially a catabolism of amino acids and presumably include several types of deamination reactions. The oxidative deamination can be written as (Savant and DeDatta, 1982):

$$\text{Amino acids} \rightarrow \text{Imino acids} \rightarrow \text{Keto acids} \rightarrow NH_3 \quad (2.11)$$

And may be operative in the oxidized soil layer. On the other hand, the reductive deamination presumably takes place in the reduced soil layer (Rose, 1976):

$$\text{Amino acids} \rightarrow \text{Saturated acids} \rightarrow NH_3 \quad (2.12)$$

Kinetically, ammonification proceeds more rapidly than nitrification (Kadlec, and Knight 1996). Ammonification occurs at all levels of soil aeration, but at varying rates, it proceeds at a much slower rate in flooded-soil system than in drained-soil system (Reddy, 1982; Reddy and Patrick, 1983). Mineralization rates are fastest in the oxygenated zone, and decrease as mineralization switches from aerobic to facultative anaerobic and obligate anaerobic microflora (Reddy and Patrick 1984). Since depth of the aerobic zone in flooded or saturated soils is usually less than 1 cm, the contribution of aerobic mineralization to the overall N mineralization would be very small, compared to facultative anaerobic and obligate anaerobic mineralization (Reddy and Graetz 1988).

The rate of ammonification in wetlands is dependent on temperature, pH value, C/N ratio of the residue, available nutrients in the system, soil conditions such as texture and structure, extracellular enzyme (such as protease), microbial biomass and soil redox conditions (Reddy and Patrick, 1984; Reddy and D'Angelo 1997). The optimum pH range for the ammonification process is between 6.5 and 8.5. In contract to most of the microbiological processes, optimum temperature for conversion of organic nitrogen to ammonium N is between 40 to 60^{o}C. However, these temperatures are seldom encountered under field conditions (Reddy and Patrick, 1984).

Ammonium N accumulation in wetland soils was found to be rapid during the first two weeks after flooding (Ponnamperuma 1972). Ammonium mineralized from fresh inputs of organic matter does not remain in the soil for long and is rapidly transferred to other N-pools in the soil/plant system through the processes of 1) volatilization, 2) nitrification, 3) plant uptake, 4) microbial immobilization, 5) ion exchange and clay fixation, 6) organic matter complexation (Killham, 1994; Paul and Clark, 1996).

Whether NH_4^+ is immobilized or accumulated in the soil depends on the microorganism's requirement of N for growth. The C:N ratio of microorganisms is not constant. Fungi can have wide C:N ratios; their C contents are quite constant at approximately 45% C. With N contents of 3 to 10%, their C:N ratios range from 15:1 to 4.5:1. Bacteria have N in their cytoplasm and in the peptidoglycan of their cell walls with C:N ratios usually in the range of 3:1 to 5:1 (Paul and Clark, 1996).

2.3.2 Ammonia volatilization

Ammonia volatilization is a physicochemical process where ammonium N is known to be in equilibrium between gaseous and hydroxyl forms as indicated by Equation (2.13):

$$NH_{3(aq)} + H_2O = NH_4^+ + OH^- \tag{2.13}$$

Reddy and Patrick (1984) pointed out that losses of NH_3 through volatilization from flooded soils and sediments are insignificant if the pH value is below 7.5 and very often losses are not serious if the pH is below 8.0. At a pH value of 8.0 approximately 95% of the ammonia nitrogen is in the form of NH_4^+ (Middlebrooks and Pano, 1983). At pH of 9.3 the ratio between ammonia and ammonium ions is 1:1 and the losses via volatilization are significant. Algal photosynthesis in wetlands as well as photosynthesis by free-floating and submerged macrophytes often creates high pH values during the day. Mikkelsen et al. (1978) reported that water pH in rice fields undergo diurnal changes, increasing by midday to values as high as pH 9.5-10 and decreasing as much as 2-3 pH units during the night. The pH of shallow flood water is greatly affected by the total respiration activity of all the heterotrophic organisms and the gross photosynthesis of the species present.

Parameters affected ammonia volatilization may be divided into several categories (Savant and DeDatta, 1982): 1) soil parameters (soil pH, salinity, alkalinity, $CaCO_3$ content, cation exchange capacity, predominant exchangeable ions, pH buffering capacity, microbial activity); 2) floodwater parameters (pH, ammonia concentration, total alkalinity, pH-buffering capacity, temperature, water movement and turbulence, depth, algal growth or activity, phosphorus concentration); 3) atmospheric parameters (air temperature, solar radiation, wind speed, partial pressure of NH_3), and 4) other parameters (nitrogen management, water management, plant canopy).

2.3.3 Nitrification

Nitrification is usually defined as the biological oxidation of ammonium to nitrate with nitrite as an intermediate in the reaction sequence. This definition has some limitations where heterotrophic microorganisms are involved but is adequate for the autotrophic and dominant species (Hauck

1984). Van de Graaf et al. (1996) defined nitrification as the biological formation of nitrate or nitrite from compounds containing reduced nitrogen with oxygen as the terminal electron acceptor. Nitrification has been typically associated with the chemoautotrophic bacteria, although it is now recognized that heterotrophic nitrification occurs (see section 2.3.3.1) and can be of significance (Keeney, 1973; Paul and Clark, 1996).

Paul and Clark (1996) pointed out that the phenomenon we now recognize as nitrification results in the production of what was historically known as saltpeter (KNO_3) and the phenomenon was first described in the Bible. In the 10th century, the saltpeter was used for production of gunpowder in China and in Europe, saltpeter plantations were described in the 16th century from Germany. Soil, manure, and lime were mixed in sheds and watered with urine and wastewater. The heaps were kept aerated, and saltpeter was extracted with hot water. It was believed that NO_3^- was formed by chemical reaction involving NH_4^+ and O_2, with the soil acting as a chemical catalyst. In the 1870s, Pasteur postulated that the formation of NO_3^- was microbiological and analogous to the conversion of alcohol to vinegar. The first experimental evidence that nitrification was biological was provided by Schloesing and Muntz in 1877. Warington in 1878, at Rothamsted, United Kingdom, found that nitrification was two-stage process involving two groups of microorganisms. One microbial group oxidized NH_4^+ to NO_2^- and another oxidized NO_2^- to NO_3^-. However, he did not isolate the responsible organisms. Winogradsky has often been credited with being the first to isolate nitrifiers. However, his paper in 1890, which assumed that the change of NH_4^+ to NO_2^- was by a single culture, was precede by one month by the publication of Franklands (1890) in Scotland. They described the isolation of a pure culture of an NH_4^+-oxidizing, NO_2^- producing bacterium that grew in inorganic media and was isolated by a dilution process (Paul and Clark, 1996).

Nitrification is generally performed by autotrophic or mixotrophic bacteria (Boettcher and Koops, 1994; Laanbroek and Woldendorp, 1995; Paul and Clark, 1996). The nitrifiers are typically aerobic bacteria although research has shown that they can have an anaerobic metabolism (Bock et al., 1988, 1995; Schmidt et al., 2001). The demonstration of an anaerobic metabolism of *Nitrosomonas europea* using pyruvate as an electron donor and NO_2^- as an electron acceptor in the presence of NH_4^+ shows the physiological diversity of these organisms (Paul and Clark, 1996). Nitrifying bacteria derive energy from the oxidation of ammonia and/or nitrite and their C is largely derived from CO_2 or carbonates (Paul and Clark, 1996). Two separate and distinct steps are involved in nitrification (Nicholas, 1963; Hauck, 1984; Bock et al., 1992; Van de Graaf et al., 1996; Verstraete and Philips, 1998; Schmidt et al., 2003).

The first step (Eq. 2.14), the oxidation of ammonium to nitrite with hydroxylamine and NO as intermediates, is mainly attributed to the genera

Nitrosomonas and *Nitrosospira* (Paul and Clark, 1996; Jetten, 2001; Schmidt et al., 2001):

$$NH_4^+ + 1.5O_2 \rightarrow NO_2^- + 2H^+ + H_2O \tag{2.14}$$

Hydroxylamine was postulated as an intermediate in ammonium oxidation already in 1926 (Kluyver and Donker, 1926). Although most physiological work has been carried out on *Nitrosomonas europea*, *Nitrosolobus, Nitrosococcus* and *Nitrosospira* have been identified as the most common NH_4^+ oxidizers in soil (Paul and Clark, 1996; Schmidt et al., 2003). The oxidation of NH_4^+ to hydroxylamine is described by Equations 2.15 and 2.16 (Killham, 1994; Paul and Clark, 1996; Hooper et al., 1997; Schmidt et al., 2001, 2003). However, Bock et al. (1991) suggested that most likely ammonia (NH_3) and not ammonium (NH_4^+) is the substrate for the oxidation process.

$$NH_3 + 2H^+ + 2e^- + O_2 \rightarrow NH_2OH + H_2O \tag{2.15}$$

$$NH_3 + N_2O_4 + 2H^+ + 2e^- \rightarrow NH_2OH + 2NO + H_2O \tag{2.16}$$

In the presence of O_2, the NO produced can be oxidized to NO_2. Therefore, only small amounts of NO are detectable in the gas phase of *Nitrosomonas* cells suspensions (Zart and Bock, 1998). While hydroxylamine is further oxidized to nitrite (Eq. 2.17), NO is (re)oxidized to NO_2 (N_2O_4) (Eq. 2.18) (Schmidt et al., 2001, 2003). Hydroxylamine is oxidized to nitrite with H_2O as the source for the second oxygen atom in nitrite (Anderson and Hooper, 1983, Killham, 1994; Paul and Clark, 1996; Jetten et al., 1997; Schmidt et al., 2003):

$$NH_2OH + H_2O \rightarrow HNO_2 + 4H^+ + 4e^- \tag{2.17}$$

$$2NO + O_2 \rightarrow 2NO_2\ (N_2O_4) \tag{2.18}$$

The second step (Eq. 2.19) in the process of nitrification, the oxidation of nitrite to nitrate, is performed by nitrite oxidizing bacteria, e.g. members of the genera *Nitrobacter*, *Nitrococcus* and *Nitrospira* (Reddy and Patrick, 1984; Prosser, 1989; Burrell et al., 1998; Hovanec et al., 1998; Schmidt et al., 2003).

$$NO_2^- + 0.5O_2 \rightarrow NO_3^- \tag{2.19}$$

giving the overall nitrification reaction (combining Eqs. 2.14 and 2.20):

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O \tag{2.20}$$

The first two genera, i.e. *Nitrobacter* and *Nitrococcus*, are part of the alpha-proteobacteria, while *Nitrospira* is phylogenetically unrelated to any other cultivated species and forms a separate division (Ehrich et al., 1995). Several strains of *Nitrobacter* and one strain of *Nitrospira* are the only nitrite oxidizers that are not restricted to marine environments (Bock et al., 1991; Watson et al., 1989). There is some evidence that *Nitrospira* is the more specialized nitrite oxidizer (Schmidt et al., 2003). The other genera are more versatile, being facultative autotrophs and anaerobes, able to grow on heterotrophic substrates such as pyruvate and also capable of the first step of denitrification, i.e. reduction of nitrate to nitrite (Bock et al., 1991).

In the case of *Nitrosomonas*, the oxidation state of nitrogen is changed from -3 to +3, and in the case of *Nitrobacter*, from +3 to +5. The energy yields to the chemoautotrophs are approximately 65 kcal (or 8.8 ATP molecules) per mole for *Nitrosomonas* and 18 kcal (or 2.5 ATP molecules) per mole for *Nitrobacter*. These energy yields are rather low compared with most heterotrophic metabolism. A mole of glucose may, for example, under optimal conditions, yield an aerobic microbe 280 kcal (or 38 ATP molecules). This partly explains why autotrophic nitrifier growth is relatively slow in the soil and even in laboratory culture where conditions for growth can be made much closer to optimal (Killham, 1994).

Nitrification is influenced by temperature, pH value, alkalinity of the water, inorganic C source, moisture, microbial population, and concentrations of ammonium-N and dissolved oxygen (Killham, 1994; Vymazal, 1995a). The optimum temperature for nitrification in pure cultures ranges from 25 to 35°C and in soils from 30 to 40°C. Nitrification, although slow below 5°C, occurs under snow cover in many soils (Paul and Clark, 1996). Optimum pH values may vary from 6.6 to 8.0. Typically, nitrification rates in agricultural soils decrease below 6.0 and become negligible below 4.5 (Paul and Clark, 1996). High pH values inhibit the transformation of NO_2^- to NO_3^-. Paul and Clark (1996) also pointed out that moisture affects the aeration regime of the soil - the water status of the soil therefore has an influence on NO_3^- production. Waterlogging limits diffusion of oxygen, and nitrification is suppressed. Nitrifying bacteria are sensitive organisms and extremely susceptible to a wide range of inhibitors including high concentrations of ammoniacal nitrogen. Approximately 4.3 mg O_2 per mg of ammoniacal nitrogen oxidized to nitrate nitrogen is needed. In the conversion process, a large amount of alkalinity is consumed, approximately 8.64 mg HCO_3^- per mg of ammoniacal nitrogen oxidized (Cooper et al. 1996).

Nitrifying bacteria have been shown to produce NO and N_2O (Bremner and Blackmer, 1978; Freney et al., 1979; Paul and Clark, 1996). Nitrite reduction in now thought to be the major process involved in these gaseous emissions, with NH_4^+ oxidation providing the electrons for the denitrification process. This process is thought to possible conserve O_2 for the ammonia mono-oxydases, keep NO_2^- from reaching toxic levels, and maintain optimum redox levels (Paul and Clark, 1996).

2.3.3.1 Heterotrophic nitrification

The involvement of heterotrophs in nitrification was first suggested in 1894 but for a long time it has been considered of little significance (Meiklejohn, 1940; Verstraete, 1975). Only recently it has been taken seriously as a soil process (Paul and Clark, 1996; Jetten et al., 1997). Heterotrophic nitrifiers are now known to be capable of producing NO_3^- from both inorganic and organic sources with the same intermediates as during autotrophic nitrification (Paul and Clark, 1996):

$$NH_4^+ \rightarrow NH_2OH \rightarrow NOH \rightarrow NO_2^- \rightarrow NO_3^- \qquad (2.21)$$

Because many of the heterotrophic nitrifiers are also denitrifiers, it is easy to underestimate the significance of this process in the nature (Paul and Clark, 1996; Focht and Verstraete 1977). Heterotrophic nitrification is often coupled with aerobic denitrification in one organism (e.g., Berks et al., 1995; Robertson et al., 1995). The oxidation of ammonium by a heterotrophic organism requires energy (contrary to autotrophic nitrification), mostly leading to decreased yield. The ecological advantage for the organism is an increased growth rate due to the simultaneous use of oxygen and nitrate as electron acceptors, which was shown for *Thiosphaera*, *Microvirgula* and other organisms (Carter et al., 1995; Robertosn et al., 1995; Patureau et al., 1998). Bacteria such as *Arthrobacter globiformis, Aerobacter aerogenes, Mycobacterium phlei, Streptomyces griseus, Paracoccus denitrificans, Alcaligenes faecalis*, *Methylococcus capsulatus*, *Thiosphaera,* and *Pseudomonas* spp. have been found to nitrify Paul and Clark, 1996; Schalk et al., 2000). The major organisms involved appear to be fungi. *Aspergillus flavus*, first isolated as a nitrifier in 1954, has been most widely studied. Species of *Penicilium* or *Cephalosporium* also are involved in nitrification (Paul and Clark, 1996). The extensive list of heterotrophic nitrifying microorganisms has been reported by Focht and Verstraete (1977).

2.3.4 Anaerobic ammonia oxidation (ANAMMOX)

Anaerobic ammonium oxidation (ANAMMOX) is the anaerobic conversion of NO_2^- and NH_4^+ to N_2. Although its existence was suggested as early as 1965 (Richards, 1965 a, b), and the existence of bacteria capable of catalyzing the ANAMMOX reaction has already been predicted by Broda in 1977 based on thermodynamic calculations, the first direct evidence for this process came from studies of activated sludge from a wastewater treatment plant in The Netherlands only much later (Van de Graaf et al., 1990; Mulder et al., 1995). It was demonstrated that in ANAMMOX process, nitrate was used as an electron acceptor. Redox balance calculations showed the following stoichometry:

$$5NH_4^+ + 3NO_3^- \rightarrow 4N_2\uparrow + 9H_2O + 2H^+ \quad (2.22)$$

During further examination of this process indications were obtained that nitrite could also serve as a suitable electron acceptor for ANAMMOX process (Van de Graaf et al., 1995):

$$NH_4^+ + NO_2^- \rightarrow N_2\uparrow + 2H_2O \quad (2.23)$$

More recently, it has become clear that nitrite is the key electron acceptor (Strous et al., 1997). The so-called ANAMMOX process is autotrophic and hence there is no need for organic substrate needed for denitrification. Furthermore, if the ANAMMOX process is combined with a preceding nitrification step, preferably blocked as nitrite, only part of the ammonium needs to be nitrified over nitrite since the ANAMMOX process combines the remaining ammonium with this nitrite to yield dinitrogen gas (Verstraete and Philips, 1998). The detailed biochemistry of the process is still under investigation in laboratory experiments and wastewater treatment plants (e.g., Jetten et al., 1999; Schalk, 2000; Schmidt et al., 2003; Strous and Jetten, 2004;), but in view of the fact that both hydroxylamine and hydrazine can act as electron acceptors, a pathway outlined in Fig. 2-6 (see also Fig. 2-4) has been proposed (Van de Graaf et al., 1996, 1997). It was proposed that hydroxylamine and ammonium are combined to yield hydrazine. Hydrazine itself is then oxidized to N_2, generating four reducing equivalents. These electrons could be used for the reduction of nitrite to hydroxylamine. The reverse of this reaction, the oxidation of ammonium to nitrite via hydroxylamine, is well known for aerobic nitrification, which occurs in both autotrophic and heterotrophic organisms (Schalk et al., 2000). It has been shown that the ANAMMOX process is strictly anaerobic and is inhibited by phosphate, oxygen and high nitrite concentrations (Strous et al., 1997, 1999a).

Recently, ANAMMOX process has also been studied in the natural environments, namely, in estuarine and offshore sediments (Dalsgaard and Thamdrup, 2002; Thamdrup and Dalsgaard, 2002; Freitag and Prosser, 2003; Trimmer et al., 2003, 2005; Meyer et al., 2005; Tal et al., 2005), in permanently anoxic bodies of water (Dalsgaard et al., 2003; Kuypers et al., 2003) and in multilayer sea ice (Rysgaard and Glud, 2004). Meyer et al. (2005) pointed out that ANAMMOX may be an important pathway in global N cycling, since it can account for as much as 67% of the benthic N_2 production, the remainder being produced by denitrification. However, the characterization of the biogeography of ANAMMOX, its significance compared to denitrification, and its regulation in nature is still incomplete, since the methods used to detect the presence and activity of ANAMMOX bacteria have become available only recently (Meyer et al., 2005; Schmid et al., 2005).

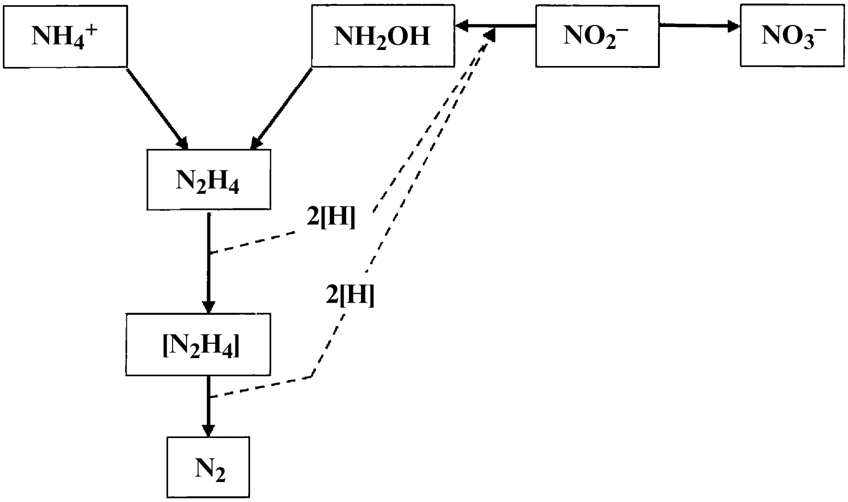


Figure 2-6. Reaction pathway for the ANAMMOX process postulated after ^{15}-N labeling studies. From Verstraete and Philips (1998), after Van de Graaf et al. (1996) with kind permission from Elsevier and Society of General Microbiology.

According to present knowledge, ANAMMOX is carried out by a few obligate anaerobic chemolitho-autotrophic bacteria that form a monophyletic cluster inside the *Planctomycetales* (Schouten et al., 2004). So far, four species have been detected from the biomass of sewage treatment plants: *Candidatus* “Brocadia anammoxidans” (Strous et al., 1999b), *Candidatus* “Kuenenia stuttgartiensis” (Schmid et al., 2000), *Candidatus* “Scalindua wagneri” and *Candidatus* “Scalindua brodae” Schmid et al., 2003). *Candidatus* “Scalindua sorokinii” was detected in the anoxic water column of the Black Sea (Kuypers et al., 2003). These bacteria have not been isolated in pure culture, and the current information about their physiology has been obtained from enrichment culture studies (Strous et al., 1999b; Egli et al., 2001). While enrichment culture studies have provided substantial insight into the physiology of the enriched species of ANAMMOX bacteria, these data may not be directly applicable to understanding of ANAMMOX in natural systems (Meyer et al., 2005). The ANAMMOX organisms so far identified in marine systems, though still *Planctomycetales*, are only distantly related to the ANAMMOX bacteria studied in enrichment cultures (Kuypers et al., 2003; Risgaard-Petersen et al., 2004).

2.3.5 Nitrate-ammonification

The first anoxic oxidation process to occur after oxygen depletion is the reduction of nitrate to molecular nitrogen or ammonia (Fewson and Nicholas, 1961). The reduction of nitrate is performed by two different groups of nitrate-reducing bacteria: the denitrifying bacteria which produce N_2O and N_2 as major reduction products (see section 2.3.6), and the nitrate-ammonifying bacteria which produce NH_4^+ as the major end product of the reduction of nitrate. In sediments and soils, both denitrification and nitrate ammonification (dissimilatory reduction of nitrate to ammonium) are observed (Keeney et al., 1971; Chen et al., 1971; Sørensen, 1978; Paul and Clark, 1996). Different numbers of electrons are used in the reduction of one molecule of nitrate in both nitrate-reducing system: 5 in the case of denitrification and 8 in the case of nitrate ammonification. Therefore, more organic matter can be oxidized per molecule of nitrate by nitrate-ammonifying bacteria than denitrifying bacteria. In addition, nitrate reduction is generally performed by fermentative bacteria that are not dependent on the presence of nitrate for growth under anaerobic conditions (Laanbroek, 1990). Hence, the carbon to nitrate ratio in a system may determine the predominance of nitrate-ammonifying or denitrifying bacteria (Tiedje et al., 1982). In marine sediments, for example, nitrate ammonification dominates at sites where excess carbon is available in relation to nitrate, whereas denitrification prevails at sites where relatively large amounts of nitrate are produced (Koike and Hattori, 1978; Sørensen, 1978; King and Nedwell, 1985, 1987). The process by which NO_3^- is reduced to NH_4^+ appears to be a dissimilatory mechanisms because it is not inhibited by NH_4^+ (Buresh and Patrick, 1978; Caskey and Tiedje, 1979).

The relative importance of denitrification and dissimilatory reduction of nitrate to ammonium in the soil environment is far from certain. Denitrification may be the dominant process in environments rich in nitrate but poor in carbon, whereas the dissimilatory reduction of nitrate and nitrite to ammonium tends to dominate in carbon-rich environments, which are preferably colonized by fermentative bacteria (Tiedje et al., 1982). So nitrate-ammonifying bacteria may be favored by nitrate-limited conditions (Laanbroek, 1990). Stanford et al. (1975) reported that the extent of NO_3^- reduction to NH_4^+ relative to denitrification increased with an increase of C/N ratio. Nitrate ammonification is found in facultative anaerobic bacteria belonging to the genera *Bacillus, Citrobacter* and *Aeromonas*, or in the members of Enterobacteriaceae (Cole and Brown, 1980; Smith and Zimmerman, 1981; Grant and Long, 1985; MacFarlane and Herbert, 1982). However, strictly anaerobic bacteria belonging to the genus

Clostridium are also able to reduce nitrate to ammonia (Hasan and Hall, 1975; Caskey and Tiedje, 1979; 1980). For many of the bacteria responsible for dissimilation to ammonium, formate is a major electron donor both for nitrate and nitrite, although most of the research on the nitrate reductase activity has been restricted to enteric bacteria such as *Escherichia coli* (Killham, 1994).

Conversion of NO_3^- to NH_4^+ and organic nitrogen increases markedly with decreasing redox potential, high pH, and large quantities of readily oxidizable organic matter (Nommik, 1956; Buresh and Patrick, 1978, 1981). Nitrate respiration to NH_4^+ occurs at Eh values of less than -100 mV (Patrick, 1960; Buresh and Patrick, 1981). Several researchers (Keeney et al., 1971; Sørensen, 1978; Koike and Hattori, 1978) have shown that dissimilatory reduction of NO_3^- to NH_4^+ can potentially occur in systems which are anaerobic for long periods while in soils where temporary anaerobic conditions exist and soils are less intensively reduced, denitrification can be the major pathway of removing nitrate.

Megonikal et al. (2004) pointed out that much less is understood about diversity and physiology of nitrate-ammonifyng bacteria than denitrifiers, despite the fact that they are sometimes the larger of the two dissimilatory NO_3^- sinks (Bonin, 1996; Bonin et al., 1998; Gilbert et al., 1997; Riviera-Monroy, 1995; Tobias et al., 2001).

2.3.6 Denitrification

The systematic study of denitrification started with workers carrying out a series of elemental analyses on soils, water and sewage, fermenting juices and manures and decaying vegetable matter (Payne, 1981). Their primitive assays first revealed the losses of nitrogen that led others, in time, to true recognition of the microbial ecology of denitrification (Reiset, 1868; Schloesing, 1868; Meusel, 1875, 1876).

The production of gaseous nitrogen compounds from nitrate in garden soil was studied by Beijerinck and coworkers (Beijerinck and Minkman, 1910), after publication on reduction of nitrate (Breal, 1892; Gayon and Dupetit, 1886). Beijerinck was the first to recognize nitrous oxide as an intermediate in denitrification. Among the bacteria capable of complete conversion of nitrate to dinitrogen gas, isolated by Beijerinck, was *Micrococcus* (now *Paracoccus*) *denitrificans*. This bacterium has become the model organism in studies on denitrification (Kluyver, 1936; Kluyver and Verhoeven, 1954; Verhoeven, 1956).

Denitrification is most commonly defined as the process in which nitrate is converted into dinitrogen via intermediates nitrite, nitric oxide and nitrous oxide (Hauck, 1984; Paul and Clark, 1996; Jetten et al., 1997). Focht and Verstraete (1977) pointed out that thought it has been well established that the initial step involves the reduction of nitrate to nitrite, considerable conflicting reports exist in much of the earlier literature concerning the intermediates between nitrite and dinitrogen. Many of these studies lacked refined equipment that did not permit detailed enzymological studies or rapid and specific identification of intermediates. However, during the 1970s, many studies (e.g., Cox and Payne, 1973 Ishaque and Aleem, 1973; Koike and Hattori, 1975) have conclusively established the following pathway, one originally proposed by Payne (1973):

$$2NO_3^- \rightarrow 2NO_2^- \rightarrow 2NO \rightarrow N_2O \rightarrow N_2 \tag{2.24}$$

The sequence of identifiable products is shown in Figure 2-7. However, the assimilatory reduction of NO_3^- to NH_4^+ and nitrification also produce N oxides (N_2O and/or NO), so that a more precise definition is desirable to keep pace with current knowledge (Hauck, 1984; Paul and Clark, 1996). From a biochemical viewpoint, denitrification is a bacterial process in which nitrogen oxides (in ionic and gaseous forms) serve as terminal electron acceptors for respiratory electron transport. Electrons are carried from an electron-donating substrate (usually, but not exclusively, organic compounds) through several carrier systems to a more oxidized N form. The

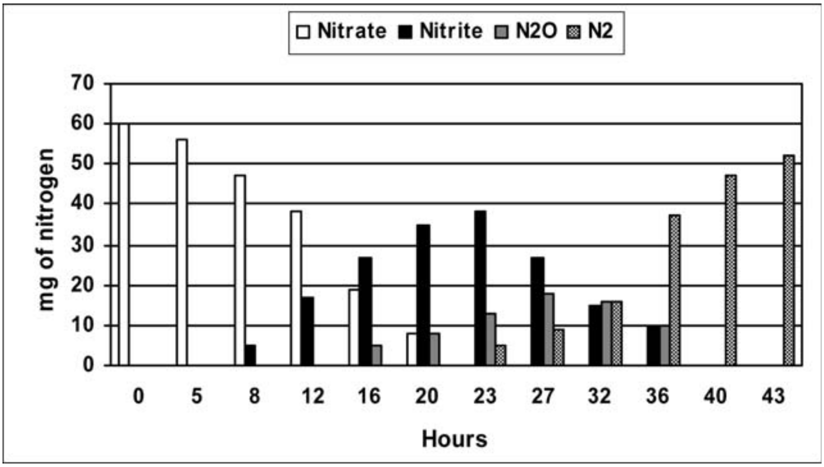


Figure 2-7. Sequential amounts of products formed during denitrification. Data from Cooper and Smith (1963).

resultant free energy is conserved in ATP, following phosphorylation, and is used by the denitrifying organisms to support respiration. Denitrification is illustrated by following equation (Hauck 1984; Reddy and Patrick, 1984):

$$5(CH_2O) + 4NO_3^- + 4H^+ \rightarrow 5CO_2 + 2N_2 + 7H_2O \tag{2.25}$$

This reaction is irreversible in nature, and occurs in the presence of available organic substrate only under anaerobic or anoxic conditions (Eh = +350 to +100 mV), where nitrogen is used as an electron acceptor in place of oxygen. More and more evidence is being provided from pure culture studies that nitrate reduction can occur in the presence of oxygen. Hence, in waterlogged soils nitrate reduction may also start before the oxygen is depleted (Kuenen and Robertson, 1987; Laanbroek, 1990).

Denitrifying bacteria are aerobes that substitute nitrate (or nitrite) for O_2 as the terminal electron acceptor when there is little or no O_2 available (Payne, 1981; Firestone, 1982). Aerobic respiration yields more free energy than NO_3^- respiration, and it is favored metabolically because O_2 inhibits key denitrification enzymes. Although some organisms can denitrify at O_2 concentrations up to 80% of the air saturation (Robertson and Kuenen, 1991), the process is most rapid *in situ* under anaerobic conditions provided there is a supply of NO_3^-. The critical concentration where denitrifiers switch to mostly anaerobic respiration is roughly ≤ 10 μmol l^{-1} (Seitzinger, 1988; Tiedje, 1988). Denitrifying activity survives well in aerobic soils, even without new enzyme synthesis or cell growth (Smith and Parsons, 1985). It persists in habitats that lack O_2 or NO_3^- presumably because denitrifiers can maintain themselves with a low level of fermentation (Jørgensen and Tiedje, 1993).

Diverse organisms are capable of denitrification. In an array are organotrophs (e.g., *Pseudomonas*, *Alcaligenes*, *Bacillus*, *Agrobacterium*, *Flavobacterium*, *Propionibacterium*, *Vibrio*), chemolithotrophs (e.g. *Thiobacillus*, *Thiomicrospira*, *Nitrosomonas*), photolithotrophs (e.g. *Rhodopseudomonas*), diazotrophs (e.g., *Rhizobium*, *Azospirillum*), archaea (e.g. *Halobacterium*) and other such as *Paracoccus* or *Neisseria* (Focht and Verstraete, 1977; Knowles, 1982; Killham, 1994; Paul and Clark, 1996). When oxygen is available, these organisms oxidize a carbohydrate substrate to CO_2 and H_2O (Reddy and Patrick, 1984). Aerobic respiration using oxygen as an electron acceptor or anaerobic respiration using nitrogen for this purpose is accomplished by the denitrifiers with the same series of electron transport systems. This facility to function both as an aerobe and as an anaerobe is of great practical importance because it enables denitrification to proceed at a significant rate soon after the onset of anoxic conditions (a redox potential of about 300 mV) without change in microbial population (Hauck, 1984). Because denitrification is carried out almost exclusively by facultative anaerobic heterotrophs that substitute oxidized N forms for O_2 as electron acceptors in respiratory processes, and because these processes follow aerobic biochemical routes, it can be misleading to refer to denitrification as an anaerobic process. It is rather one that takes place under anoxic conditions (Hauck, 1984).

Some species of *Propionibacterium* that denitrify are obligately fermentative; in *Bacillus*, some are facultatively fermentative. Even S can be used as an electron source. Under anaerobic conditions *Thiobacillus denitrificans* oxidizes elemental sulfur and reduces nitrate as follows (Paul and Clark, 1996; Zhang and Lampe, 1999):

$$5S + 6KNO_3 + 2H_2O \rightarrow 3N_2 + K_2SO_4 + 4KHSO_4 \tag{2.26}$$

Many environmental factors are known to influence denitrification rates including degree of aeration, redox potential, soil moisture, temperature, pH value, presence of denitrifiers, soil type, organic matter, nitrate concentration and the presence of overlying water (e.g. Focht and Verstraete, 1977; Payne, 1981; Knowles,1982; Vymazal, 1995a).

As most denitrification is accomplished by heterotrophic bacteria, the process is strongly dependent on carbon availability. There is a general correlation between total soluble organic matter content and denitrification potential, but much better correlation occurs with the supply of easily decomposable organic matter or water-extractable organic carbon (Bremner and Shaw, 1958; Broadbent and Clark, 1965; Burford and Bremner, 1975; Paul and Clark, 1996). Organic substances able to act as sources of energy and as hydrogen donors may be present in sediments and soils through the decomposition of tissues or be provided by living roots exudates (Bailey, 1976; Stefanson, 1973). The amount of water-soluble C measurable in soil has been found to account for 71% of its denitrification potential, and the amount of C mineralizable during 7 days of incubation was more than adequate to account for all of the requirements (Paul and Clark, 1996).

The distribution of denitrifying bacteria covers a much wider range of the pH spectrum that the autotrophic nitrifying bacteria (Focht and Verstraete, 1977). Though the process is favored at slightly alkaline pH (Wiljer and Delwiche, 1954; Nommik, 1956; Bremner and Shaw, 1958; Dawson and Murphy, 1972), it proceeds to a pH of 3.5 (Nommik, 1956; Cady and Bartholomew, 1960). On the other hand, denitrification may occur in wastes up to about pH 11 (Prakasam and Loehr, 1972). Most of the common soil and aquatic denitrifying bacteria have optima for growth similar to those of the general bacterial flora, i.e. at least in the pH 5-9 range (Focht and Verstraete, 1977) with majority of nitrifiers having an optimum in the range of 7.0 to 8.0 (Wijler and Delwiche, 1954; Nommik, 1956; Van Cleemput and Patrick, 1974; Delwiche and Bryan, 1976; Müller et al., 1980).

Denitrification is also strongly temperature dependent. Rates of denitrification increase up to a maximum in the region of 60 to 75^{o}C and then decline rapidly above this temperature (Bremner and Shaw, 1958; Keeney et al., 1979; Knowles, 1982; Paul and Clark 1996). It is usually assumed that enrichment with thermophillic bacilli (Verhoeven, 1952) or other bacteria is responsible for the observed activity at high temperature, but it is speculated that abiological reactions may also be important (Keeney et al., 1979). Denitrification proceeds at very slow but measurable rates, at temperatures below 5°C (Bremner and Shaw, 1958; Smid and Beauchamp, 1976) where relatively large mole fractions of N_2O and NO are reported. Molecular N_2 is the chief product at higher temperatures (Broadbent and Clark, 1965). Christensen and Sørensen (1986) reported that light conditions inhibited denitrification in both the surface layer and the upper part of the root zone, suggesting that a release of oxygen by benthic algae controlled a diel variation of denitrification.

The soil water content regulates oxygen availability in soil and thus denitrification (Kaplan et al., 1979; Paul and Clark, 1996). Roughly, half the variation in denitrification rates in the field can be explained by soil moisture. The relationship between water-filled pore space (WFPS) and denitrification is shown in Figure 2-8. This shows the onset of denitrification occurring at about 60% WFPS. The occurrence of both denitrification and nitrification in the 60-90% WFPS range may variously involve the occurrence of highly individualistic microsites in soil of the presence of organisms with both denitrifying and nitrifying capability (Paul and Clark, 1996).

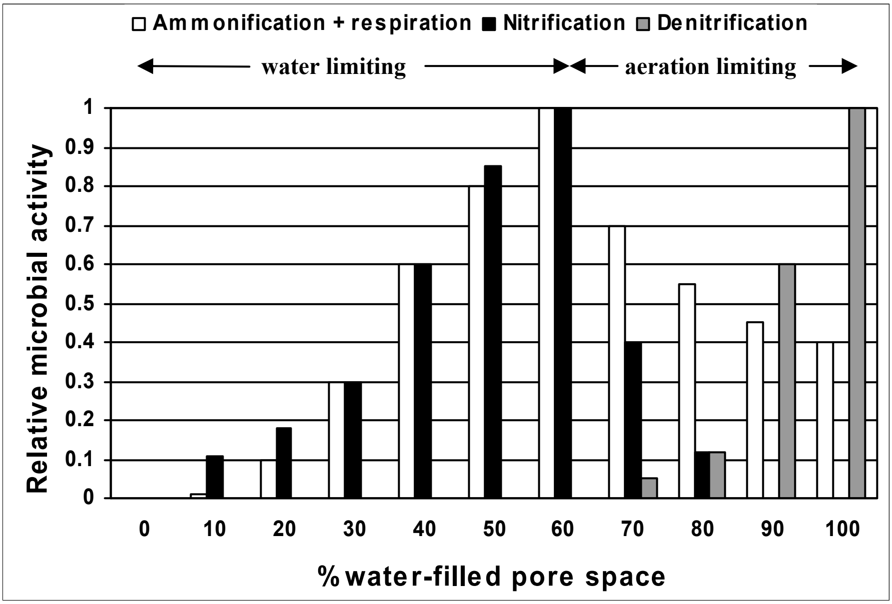


Figure 2-8. Relative rate soil denitrification and other microbial processes as function of percentage water-filled pore space. Data from Linn and Doran (1984).

The quantity of N_2O evolved during denitrification depends upon the amount of nitrogen denitrified and the ratio of N_2 to N_2O produced. The ratio is also affected by aeration, pH value, temperature, and nitrate to ammonia ratio in the denitrifying system. Blackmer and Bremner (1979) reported that the concentration of NO_3^- has two effects on N_2O reduction by soil microorganisms, one being that it stimulates production of N_2O reductase, the other being that it inhibits the activity of this enzyme. Firestone et al. (1979) reported that as the concentration of NO_2^- increases the proportion of N_2O occurring as the gaseous product of denitrification increases. This is not due to an increase in rate of N_2O production, for the total rates of $N_2O + N_2$ production do not vary significantly over the range of NO_2^- concentration used. The authors pointed out that the net result of an increase in NO_2^- concentration must be a decrease in the rate of N_2O reduction to N_2.

Carlson and Inghram (1983) reported that although *Pseudomonas stutzeri*, *Pseudomonas aeruginosa* and *Paracoccus denitrificans* reduced to dinitrogen gas, they did so at different rates and accumulated different kinds and amounts of intermediates. Their rates of anaerobic growth on nitrate varied about 1.5-fold; concomitant gas production varied more than 8-fold.

The $N_2O : N_2$ ratio is also affected by aeration, pH, temperature and nitrate to ammonia ratio in the denitrifying system (Vymazal, 1995a). If the pH is below 4.5, the denitrification rate is relatively slow and only N_2O is released; at pH > 5, N_2 is the main end product of denitrification under conditions of low redox potential, whereas the relative importance of N_2 decreases if the circumstances are less anaerobic (Armentano and Verhoeven, 1991). In oxygen-depleted sediments and soils, where the pH is > 6, the rate of conversion of N_2O to N_2 increases faster than the rate of formation N_2O (Focht, 1974). The rate of N_2O emission increases with increasing moisture content of the soil and with increasing temperature up to 37°C.

Nitrification and denitrification are known to occur simultaneously in flooded soils and sediments where both aerobic and anaerobic zones exist (Greenwood, 1962; Patrick and Reddy, 1976; Reddy and Patrick, 1986). Reddy and Patrick (1986) pointed out that the significance of nitrification-denitrification reactions in flooded lowland soils had been recognized as early in 1935 and then confirmed in the 1950s (e.g., Pearsall, 1950) Flooded soil or sediments containing an aerobic surface layer over an anaerobic layer or the aerobic root rhizosphere of a wetland plant growing in an anaerobic soil could be the examples. By combining these two reactions, a balanced equation occurring in aerobic and anaerobic layers can be written as (Reddy and Patrick, 1984):

$$4NH_4^+ + 8O_2 \rightarrow 4NO_3^- + 4H_2O + 8H^+ \tag{2.27}$$

$$4NO_3^- + 5/6C_6H_{12}O_6 + 4H^+ \rightarrow 2N_2 + 5CO_2 + 7H_2O \tag{2.28}$$

$$4NH_4^+ + 5/6C_6H_{12}O_6 + 8O_2 \rightarrow 2N_2 + 5CO_2 + 11H_2O + 4H^+ \tag{2.29}$$

These reactions involve both oxidation (nitrification) and reduction (denitrification), with a change of valence of N from -3 (for NH_4^+) to +5 (for NO_3^-), followed by a reduction to +1 (for N_2O) and zero (for N_2). Four moles of oxygen are required to react with 2 moles of ammonium to produce one mol of N_2 gas.

2.3.6.1 Aerobic denitrification

Nitrite reduction to gaseous products by denitrifying bacteria used to be considered to be a strictly anaerobic process, but this fallacy was dispelled with the discovery of aerobic denitrification (Robertson and Kuenen, 1984). There is no fundamental argument why denitrification cannot occur under oxic conditions (Jetten, 2001). However, only during past few years has this activity received some attention (Robertson and Kuenen, 1990; Robertson et al., 1995; Gupta, 1997; Stouthamer et al., 1997; Patureau et al., 1998; Scholten et al., 1999). Furthermore, the name aerobic denitrification is used in different contexts, which leads to additional confusion. It is mostly used to refer to microorganisms, which denitrify while sensing oxygen, but in some cases it is used to refer to denitrification in an oxic system. In the latter case diffusion limitation in flocs, biofilms, soil or not well mixed systems provides anoxic pockets where conventional denitrification can take place (Jetten, 2001).

Aerobic denitrification is often coupled to heterotrophic nitrification in one organism (see section 2.3.3.1). Because nitrification is mostly measured by the formation of nitrate or nitrite under oxic conditions, while (aerobic) denitrification is not expected under such conditions, this coupled process is not easily observed in standard enrichment cultures. The observation that *Thiosphaera pantotropha* and other organisms are not only heterotrophic nitrifiers, but also aerobic denitrifiers forced a re-evaluation of this approach (Ludwig et al., 1993; Baker et al., 1995; Jetten, 2001). Aerobic denitrifiers are present in high number in natural soil samples. Even though the specific activities are not always very high, they are sufficient to allow significant contribution to the turnover of compounds in the nitrogen cycle (Jetten et al., 1997).

2.3.7 Fixation

Nitrogen fixation has been shown to be a significant process in submerged soils. Organic N tends to accumulate in submerged soils and wetlands in higher concentration that in drained soil. Conversion of drained land areas to flooded conditions invariably results in higher N content, while the reverse process results in loss of N (Buresh et al., 1980a).

Nitrogen fixation is the conversion of gaseous nitrogen (N_2) to ammonia. Nitrogen fixation requires nitrogenase, an oxygen-sensitive iron-, sulfur- and molybdenum-containing enzyme complex which also brings about the reduction of other substrates containing triple covalent bonds (e.g., nitrous oxide, cyanides or acetylene). Nitrogen-fixing enzyme nitrogenase is very sensitive to inactivation by oxygen; similar to the nitrogenase of the bacterial anaerobe *Clostridium*, where activity is lost in air, but dissimilar to the nitrogenase of the aerobic bacterium *Azotobacter*, where the nitrogenase is stable in air (Stewart, 1973).

The reduction of gaseous nitrogen (N_2) to ammonia (NH_3) takes place very rapidly and for this reason the individual steps in the reaction have not been investigated in detail. It is supposed that the whole reaction is a 3-step, 2 electrons per step mechanism (Winter and Burris, 1976):

$$\underset{}{N{\equiv}N} \Rightarrow \underset{\text{diimide}}{HN{=}NH} \Rightarrow \underset{\text{hydrazine}}{H_2N\text{-}N_2H} \Rightarrow NH_3 \tag{2.30}$$

In wetland soils, biological N_2 fixation may occur in the floodwater, on the soil surface, in aerobic and anaerobic flooded soils, in the root zone of plants, and on the leaf and stem surfaces of plants (Buresh et al., 1980a). A wide variety of symbiotic (associated with nodulated host plants) and asymbiotic (free-living) organisms can fix nitrogen in wetlands (Johnston, 1991).

There are six main types of N_2-fixing organisms that can be found in soil (Killham, 1994): 1) free-living bacteria such as *Bacillus*, *Klebsiella* and *Clostridium* that fix N_2 anaerobically (the first two are facultative anaerobes and fix nitrogen under reduced oxygen tensions while *Clostridium* is an obligate anaerobe); 2) bacteria of the genus *Rhizobium*, which fix N_2 mainly in the root nodules of leguminous plants; 3) actinomycetes of the genus *Frankia*, which fix N_2 in the root nodules of non-legimunous angiosperms such as *Alnus glutinosa* (those associations are often referred to as "actinorhizas"); 4) free living cyanobacteria on the soil surface such as *Nostoc* and *Anabaena*; 5) symbiotic cyanobacteria found in the lichen symbiosis; and 6) N_2-fixing bacteria loosely associated with the roots of certain plants, sometimes referred to as "rhizocoenoses" (e.g. *Azotobacter*, *Beijerinckia* and *Azospirilllum*). In wetland systems, free-living bacteria, cyanobacteria (blue-green algae), N_2-fixing bacteria loosely associated with the roots of certain plants and probably *Frankia* are the most important N_2-fixing organisms.

Paul and Clark (1996) pointed out that the significance of nodules in symbiotic N_2 fixation was described by Helriegel and Wilfarth in 1886. Beijerinck, in 1888, isolated the organisms responsible for N_2 fixation by legumes and called them *Bacillus radiciola*. These were later reassigned to *Rhizobium*. Asymbiotic N_2-fixing anaerobes in the genus *Clostridium* were isolated by Winogradsky in 1890, and asymbiotic aerobe (*Azotobacter chroococcum*) was described by Beijerinck in 1901.

2.3.7.1 Free-living bacteria

Nitrogen-fixing activity associated with free-living heterotrophic bacteria has been reported in freshwater (Howard et al., 1970; MacGregor et al., 1973, Paerl et al., 1981), freshwater marshes (DeLaune et al., 1986; Sprent, 1987), estuarine (Brooks et al., 1971; Marsho et al., 1975), and salt marsh sediments (Teal et al., 1979; Whiting et al., 1986).

The best know aerobic N_2-fixing bacteria are members of the genus *Azotobacter*. *A. chroococcum* was the second free-living N_2-fixing microbe to be discovered by Dutch microbiologist W.W. Beijerinck in 1901, after Winogradsky's *Clostridium pasteurianum* in 1893 (Postgate, 1978).

The aerobic, free-living, N_2-fixing bacteria that utilize organic, often recalcitrant substrates as a source of energy include *Azotobacter*, found in neutral and alkaline soils. Members of the same family, *Beijerinckia* and *Derxia*, have a broader pH range and are more often found in acidic soils, especially in the tropics. Genetic analysis shows *Beijerinckia* to be more closely related to another N_2-fixing bacterium, *Azospirillum*, than to *Azotobacter*. These organisms, as well as N_2-fixing *Enterobacter*, grow in humid environments on leaf surfaces or in leaf sheaths, as well as in the soil and on root surfaces (Paul and Clark, 1996).

Azotobacter, *Beijerinckia*, and *Rhizobium* require aerobic conditions for the production of the extensive energy required for N_2 fixation. However, in these organisms, as in all other diazotrophs, the activity of nitrogenase is inhibited by oxygen. Special mechanisms for the protection of nitrogenase include the association of the N_2-fixing complex with membranes within the cell, slime production, and clump formation. Another feature of aerobic N_2-fixing bacteria is the high level of respiration within the cell. This in *Azotobacter* helps protect the enzyme from O_2 by maintaining low O_2 concentrations (Paul and Clark, 1996).

Nitrogen fixation in flooded soils and sediments has been predominantly attributed to the anaerobe *Clostridium* (Rice et al., 1967, Brooks et al., 1971, Rice and Paul, 1972) but other anaerobes, such as *Desulfovibrio* (Herbert, 1975), *Desulfotomaculum* (Postgate, 1978; Paul and Clark, 1996) may also be important. These organisms use organic compounds as electron donors. The fermentative pathways of these organisms lead to the build-up of organic intermediates and result in low amounts of energy being available for N_2 fixation. However, certain environmental conditions with high substrate availability combined with anaerobic conditions such as waterlogging, result in extensive nitrogen fixation (Paul and Clark, 1996).

Facultative microaerophilic organisms such as *Klebsiella*, *Azospirillum*, *Enterobacter, Citrobacter, Escherichia Propionibacterium*, and *Bacillus* (Postgate, 1978; Paul and Clark, 1996) produce energy in the form of ATP by oxidative pathways in an environment where nitrogenase does not need to be as well protected from oxygen (Paul and Clark, 1996). These organisms are a physiological group of bacteria which are able to grow either with or without the oxygen, but which can only fix nitrogen anaerobically. Therefore, it took many years before their ability to fix nitrogen became firmly established (Postgate, 1978).

Also many phototrophs which make use of light to fix CO_2 are able to fix nitrogen. The strict anaerobes such as colored bacteria *Chromatium* (red or purple) and *Chlorobium* (green) are sulfur bacteria which oxidize sulfides, elemental sulfur or thiosulfates to sulfates while absorbing light with chlorophyll and carotenoid pigments. The energy is used for growth and can be coupled to N_2-fixation. The colored N_2-fixing, non-sulfur bacteria such as *Rhodospirillum rubrum* and *Rhodopseudomonas palustris* (both purple) grow phototrophically, but only anaerobically (Postgate, 1978). Wetzel (2001) summarized that nitrogen-fixing photosynthetic bacteria can fix limited quantities of N_2 in the dark, but at rates usually $< 10\%$ of maximum daytime rates (Horne, 1979; Levine and Lewis, 1984).

2.3.7.2 Cyanobacteria

Nitrogen fixation by cyanobacteria was first reported in 1889, shortly after the significance of legume root nodules was first shown (Sprent and Sprent, 1990). Cyanobacteria (blue-green algae) were shown by Drewes in 1928 to fix N_2. They occur in many aquatic situations. They also are found at or just below the surface of many soils, especially those that have been newly exposed by erosion or landslides. They are only prokaryotes that exhibit photosynthetic mechanisms characteristic of higher plants, in that they contain photosystem II and evolve O_2, yet they can survive in low-light, low-O_2 environments (Paul and Clark, 1996). Under anaerobic conditions heterocysts are not formed and nitrogen fixation proceeds in vegetative cells of cyanobacteria (Vymazal, 1995a).

The free-living, diazotrophic cyanobacteria have three morphological forms. The first is unicellular and the other two, filamentous. The unicellular cyanobacteria belong to the family Chroococcaceae, for example, genera *Gloeothece* and *Synechococcus*. In this case, most N_2 fixation occurs in the dark (Sprent and Sprent, 1990; Paul and Clark, 1996). The second morphological group includes filamentous, heterocystous cyanobacteria in which the ability to fix nitrogen is especially favored in specialized cells called heterocysts (Stewart, 1969, 1975, 1977; Wolk, 1973; Fogg, 1974; Riddolls, 1985; Haselkorn and Buikema, 1992). The thick-walled heterocysts which occur every 10 to 15 cells in the filament, contain the nitrogenase and protect it from oxygen damage by membranes that help control N_2 and O_2 diffusion rates relative to their utilization Paul and Clark, 1996). Cyanobacteria in this group belong, for example, to the families Nostocaceae, Scytonemataceae, Stigonemataceae (Sprent and Sprent, 1990; Paul and Clark, 1996). Probably the most widely studied species in this group is *Anabaena cylindrica*, which has ever been referred to as the "*E. coli*" of cyanobacteria (Sprent and Sprent, 1990). The third morphological group includes filamentous non-heterosystous cyanobacteria in the family Oscillatoriaceae. These differ from *Gloeothece* in that N_2 fixation is light associated when O_2 also is produced.

2.3.7.3 N_2-fixing bacteria loosely associated with the roots

The rhizosphere was shown many years ago to be an active site for N_2 inputs. Döbereiner et al. (1972) and Döbereiner and Day (1975) reported that the association of *Azotobacter paspali* with the roots of tropical grass, *Paspalum notatum*, fixed en estimated 9 g N m^{-2} yr^{-1}. Later research rediscovered the previously described diazotrophic spirilli now named *Azospirillum*. The genus is associated with a diverse range of plant hosts, such as the salt marsh grass *Spartina* (Paul and Clark, 1996). De Polli et al. (1977) confirmed both N_2 fixation by bacteria associated with the roots of tropical grasses and plant assimilation of the fixed N by $^{15}N_2$ reduction method. Buresh et al. (1980a) summarized that non-symbiotic nitrogen fixation by bacteria has also been found in the rhizosphere of rice (Rinaudo et al., 1971; Yoshida and Ancajas, 1973; Dommergues et al., 1973), freshwater (Bristow, 1974; Silver and Jump, 1975), marine (Patriquin and Knowles, 1972) and salt marsh angiosperms (Jones, 1974; Patriquin and McClung, 1978).

The soil-root interface can be divided into three regions: 1) the outer rhizosphere comprising the soil immediately surrounding the root, 2) the rhizosphere or actual root surface, and 3) the inner rhizosphere (also called the endorhizosphere or histosphere) comprising the cortical tissue of the roots. Each region contains heterotrophic bacteria, some of which are able to fix nitrogen (Buresh et al., 1980a). Larger populations of N-fixing bacteria are observed in rhizosphere than non-rhizosphere soil (Balandreau et al., 1975; Patriquin and Knowles, 1972). Nitrogenase activity in a salt marsh has been shown to approximate the root distribution of *Spartina alterniflora* (Hanson, 1977; Casselman et al., 1981).

2.3.7.4 Factors affecting nitrogen fixation

A number of environmental factors influence the rate of nitrogen fixation in flooded soils The availability and quality of carbon compounds appear to be the primary factor limiting growth of heterotrophic nitrogen-fixing bacteria because these microorganisms must obtain their energy from carbon compounds synthetized by other organisms. Added labile carbon substances have been shown to stimulate nitrogen fixation in flooded soils and excretion of organic compounds from plant roots helps make the rhizosphere a favorable environments for heterotrophic nitrogen fixation (Buresh et al. 1980a). It has also been speculated that plant photosynthesis promotes root exudation which in turn acts as a source of carbon and energy for heterotrophic nitrogen fixers in the root zone (e.g. Yoshida and Ancajas, 1973; Basilier et al., 1978; Zuberer and Silver, 1978).

Nitrogen fixation has also been correlated positively with concentrations of dissolved organic nitrogen occurring in the water (Horne and Fogg, 1970; Paerl, 1985) and total phosphorus concentrations (Lundgren, 1978; Brownlee and Murphy, 1983; Wurtsbaugh et al., 1985). Factors that inhibit nitrogen fixation include high ambient nitrogen concentration of inorganic nitrogen, low light intensities (decreases autotrophic nitrogen fixation), high oxygen concentrations (inhibits nitrogenase), high redox potential (fixation is greater under reduced than under oxidized conditions), and high (>8.0) or low (< 5.0) pH levels (Buresh et al., 1980a).

Light-dependent nitrogen-fixing activity in flooded soil systems is attributed mostly to cyanobacteria and in some cases to photosynthetic bacteria. Photosynthetic nitrogen fixers are confined to the photic zone of flooded soils which includes the column of water overlying the soil, the surface of the soil layer, and the leaf and stem surface of plants (Buresh et al. 1980). Nitrogen fixation decreases with depth down the soil profile in estuarine, freshwater as well as salt marsh soils (Brooks et al., 1971; Keirn and Brezonik, 1971; Marsho et al., 1975; Whitney et al., 1975). Nitrogen-fixing photosynthetic bacteria require anaerobic conditions for growth (Buresh et al., 1980a). Nitrogen fixation within the soil layer of flooded soil systems is greater under reduced than under oxidized conditions. It has been reported that greatest nitrogen-fixing activity in a flooded soils was found within the redox range from -200 to -260 mV (Buresh et al., 1980a).

Although N_2-fixation by free-living bacteria in unamended soils is low compared to fixation by cyanobacteria, bacteria occupy a larger fraction of the flooded habitat than cyanobacteria and over a period the amount fixed by bacteria may be significant (Jones, 1974; Whitney et al., 1975; Carpenter et al., 1978; Teal et al., 1979). Availability of an energy source appears to be the primary factor limiting nonphotosynthetic N_2-fixing bacteria (Stewart, 1969; Hanson, 1977).

2.3.8 Plant uptake and assimilation

Nitrogen assimilation refers to a variety of biological processes that convert inorganic nitrogen forms into organic compounds that serve as building blocks for cells and tissues (Kadlec and Knight, 1996). Macrophyte growth is not the only potential biological assimilation process: microorganisms and algae also utilize nitrogen. Ammonia is readily incorporated into amino acids by many autotrophs and microbial heterotrophs (Vymazal, 1995a).

Nitrogen can be absorbed by plants in three distinct forms: nitrate, ammonium and amino acids. There is a substantial carbon cost in nitrogen assimilation, i.e., conversion of inorganic to organic nitrogen. Nitrate must be first reduced to ammonium, which must be then attached to a carbon skeleton before it can be used in biosynthesis. Thus the carbon cost of assimilation, which is generally large, is nitrate > ammonium > amino acids (Clarkson, 1985).

The two forms of nitrogen generally used for assimilation are NH_4^+ and NO_3^-. Almost all green plants, fungi and bacteria can reduce NO_3^- to NH_4^+ in the course of biosynthesis of amino acids and proteins. The biochemical pathway is (Paul and Clark, 1996):

$$NO_3^- \rightarrow NO_2^- \rightarrow NH_2OH \rightarrow NH_4^+ \rightarrow R\text{-}NH_2 \tag{2.31}$$

The first three reductions duplicate those accomplished by certain bacteria during dissimilatory reduction of NO_3^- to NH_4^+. Assimilative reductases are soluble proteins and are repressed by NH_4^+. Some bacteria lack NO_3^- reductase but contain NO_2^- reductase. Possibly, this is a defensive adaptation as NO_2^- can be mildly toxic (Paul and Clark 1996).

Depending on the species, nitrate is reduced either in the roots or transported to the leaves, where it is reduced in the light. The first step in the reduction is catalyzed by nitrate reductase, which is an inducible enzyme; the gene encoding nitrate reductase is transcribed in response to nitrate application (Campbell, 1996). The protein is rather short-lived, being degraded with a half-time of a few hours (Li and Oaks, 1993). Although shoot nitrate reductase activity is energetically advantageous due to the direct coupling with photosynthesis, high nitrate reductase of roots compared to shoots was observed in different submerged wetland species (Cedergreen and Madsen, 2003) or plants colonizing acid soils with prevailing NH_4^+-N source (Claussen and Lenz, 1999). Munzarova et al. (2006) pointed out that in spite of the fact that root NO_3^- reduction may depend on anaerobic metabolism of carbohydrates, oxygen shortage induced nitrate reductase activity (Garcia-Novo and Crawford, 1973; Muller et al., 1994; de la Haba et al., 2001) or increased the relative importance of root nitrate reductase activity to whole plant nitrate reductase activity (Jiang and Hull, 1999).

Munzarova et al. (2006) reported that the nitrate reductase does not necessarily correspond with plant ability to take up NO_3^- and grow under NO_3^--N source. The authors found that *Phragmites australis* and *Glyceria maxima* differed significantly in the content of compounds interfering with nitrate reductase activity estimation. *Glyceria*, but not *Phragmites*, contained cyanogenic glycosides releasing cyanide, the potent nitrate reductase inhibitor. They concluded that the use of nitrate reductase activity as a marker for NO_3^- utilization in individual plant species is impossible without the precise method optimization.

With both NH_4^+ and NO_3^- present in the soil or growth substrate, the ionic source for N generally preferred by organisms for protein synthesis is NH_4^+. Nitrogen in the reduced form can be incorporated readily into amino acids whereas the oxidized form must first undergo reduction (Vymazal, 1995a). Nitrate assimilation is energetically expensive because of the costs of nitrate reduction. Ammonium is toxic to plant cells and therefore must be assimilated rapidly to amino acids (Lambers et al., 1998). Many plant species show reduced growth under strict NH_4^+ nutrition and develop NH_4^+ toxicity syndrome, which is associated with an accumulation of NH_4^+ in tissues (Mehrer and Mohr, 1989). In contrast, NO_3^- can be stored in vacuoles without detrimental effects and the ion participates in osmoregulations (Marschner, 1995). Rough calculations suggest that nitrate reduction to ammonium requires approximately 15% of plant-available energy when it occurs in the roots (2% in plants that reduce nitrate in leaves), with additional 2 to 5% of available energy for ammonium assimilation (Bloom et al., 1992).

Plant species differ in their preferred forms of nitrogen absorbed, depending on the forms available in the soil (Lambers et al., 1998). Most plants, however, are capable of absorbing any form of soluble nitrogen, especially if acclimated to its presence (Atkin, 1996). The preference for a particular ion is an important factor affecting plant community composition (Kronzucker et al., 1997). The NH_4^+ preference is common in plant occupying habitats with restricted nitrification, where NH_4^+ prevails (Kronzucker et al., 1997; Garnett et al., 2001). NH_4^+ dominates in waterlogged sediments since NO_3^- decrease rapidly with soil depth (Andersen and Hansen, 1982) and is limited to hypoxic top soil layer and oxidized patches around roots of emergent plants. Nutrient enrichment enhances plant productivity, litter accumulation and microbial activities, facilitating O_2 demand and diminishing the NO_3^- availability in the sediment (Kühl et al., 1997; Nijburg and Laanbroek, 1997; Čížková et al., 2001).

Tylova-Munzarova et al. (2005) pointed out that wetland plants are supposed to favor NH_4^+ over NO_3^-. However, among speciess occupying waterlogged soils, the preferential uptake of NH_4^+ is documented for rice (*Oryza sativa*) only (Sasakawa and Yamamoto, 1978), and both its uptake and assimilation is enhanced by the presence of NO_3^- (Kronzucker et al., 1999).

Nutrients are assimilated from the sediments by emergent and rooted floating-leaved macrophytes, and from the water in the free-floating macrophytes (Wetzel, 2001). Various experiments have proven that minerals can be taken up directly by shoots of submerged plants. However, there is also no question regarding the uptake capacity of nutrients by the roots of these plants (Vymazal, 1995a). The ability of rooted macrophytes to utilize sediment nutrients may partially account for their greater productivity in comparison with planktonic algae in many systems (Wetzel, 2001).

Nitrogen is taken up and assimilated by growing plants throughout the growing season. However, uptake rate varies widely during the growing season. The uptake rates are much higher during the exponential growth phase as compared to steady state growth phase. For example, Okurut (2001) reported that nitrogen uptake rates for *Cyperus papyrus* and *Phragmites* *mauritianus* growing in a constructed wetland in Uganda during the exponential and steady state growth phases were 0.71 and 0.03 g N m^{-2} d^{-1}, respectively. Therefore, it could be very misleading to estimate annual uptake based on the short-term uptake experiments.

The highest concentrations in live plant tissue occur early in the growing season and decrease as plants mature and senesce. This trend has been documented many times in the literature (e.g., Boyd, 1970a; Dykyjová and Hradecká, 1976; Behrends et al., 1994; Vymazal, 2005a; Fig. 2-9). As the rate of biomass and nutrient accumulation diminishes, translocation of nutrients and photoassimilate from leaves to rhizome occurs. Patterns of seasonal changes in composition vary for both species and nutrient and broad generalization probably cannot be made. The seasonal changes are also influenced by many environmental factors and may reach even more than 50% (Dykyjová and Hradecká, 1976; Bayley and Freeman, 1977; Kufel, 1982; Gopal, 1990; Vymazal, 1995a). For example, Garver et al. (1988) studied in detail seasonal patterns in accumulation and partitioning of biomass and macronutrients in *Typha* spp. grown in cultivated stands in Minnesota. They reported that biomass and nutrient accumulation are in a lag phase during the first 4-8 weeks of growth in the spring. The plants then enter a rapid growth phase in which 47-80% of the total seasonal biomass production and nutrient uptake occurs in a 4-8 week period. During this time, *Typha* leaves account for 60-70% of the biomass and represent the major nutrient sink. As the rate of biomass and nutrient accumulation diminishes, translocation of nutrients and photoassimilate from leaves to rhizomes occurs, with the estimated 40% of leaf nitrogen, 35-44% of leaf phosphorus and 4-38% of leaf potassium translocated to the rhizomes by the end of October. Over the winter, 75% or more of the rhizome biomass, N, P, and K is preserved. Gopal and Sharma (1984) estimated that about 35%N and 50%P of the total maximum content in the leaves of *Typha elephantina* were translocated to the rhizomes during winter and later some of it is used again for new shoot growth.

The content of mineral nutrients in macrophytes is controlled by both the habitat and the interspecific variations of the plants. Also, the developmental rate of various species must be take into consideration. In some species, such as *Acorus calamus* (Sweet flag) the vegetative phase of the aboveground parts is very short and in temperate regions its reproductive phenophase is over in July. After having attained its maximum biomass the shoots become senescent. On the other hand, species as *Glyceria maxima* (Mannagrass) begin to sprout in very early spring and the formation and growth of new tillers continue during the whole growing season, sometimes even during the mild winter period (Dykyjová, 1973).

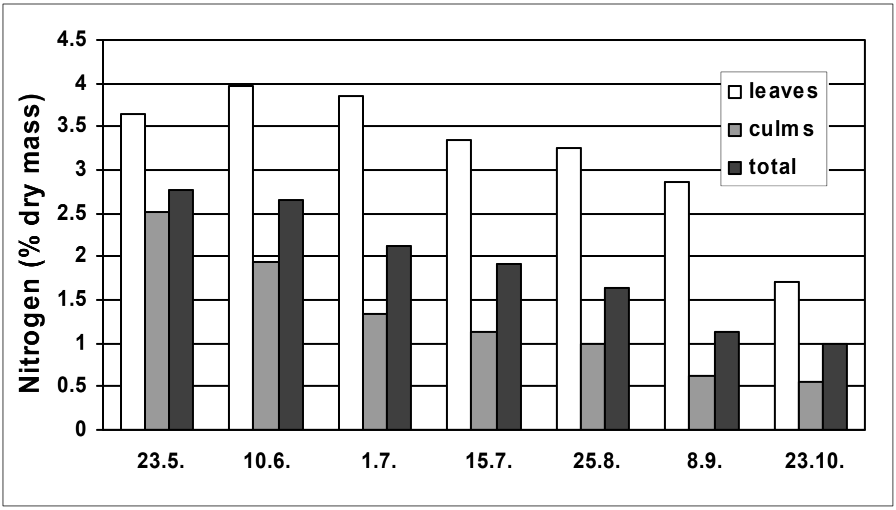


Figure 2-9. Concentration of nitrogen in various parts of *Phragmites australis* from littoral of the Rožmberk Pond in south Bohemia during the growing season. Data from Dykyjová and Hradecká (1976).

Total storage of a substance in a particular compartment is called standing stock. Nutrient standing stocks in the vegetation are commonly computed by multiplying nutrient concentrations in the plant tissue by biomass per unit area and are expressed as mass per unit area (usually g m^{-2} or kg ha^{-1}) (Johnston, 1991). It implies that nutrient standing stocks in live biomass depend on both nutrient concentrations in the plant tissue as well as on the amount of live plant biomass. However, it is well known that peak nutrient concentration and peak live biomass do not occur at the same time of the growing season in northern temperate latitudes. Maximum nutrient concentrations are found early in the growing season while peak live aboveground biomass occurs later in the growing season (in temperate climate usually in the period of July-August depending on the species, at lower latitudes aboveground standing stocks tend to peak earlier). The major factor influencing the nutrient standing stock is the biomass. While the nutrient concentrations at the time of peak biomass may be 50% lower as compared to beginning of the season (e.g., April, May), the live biomass dry matter during the peak biomass may be 10 - 30 times higher for some species (Richardson and Vymazal, 2001). Therefore, the peak nutrient standing stock in northern temperate latitudes usually occurs at the time of peak live biomass or shortly before that, and decline during autumn senescence (Fig. 2-10).

Most of the biomass, and its contained nitrogen, decomposes to release carbon and nitrogen. Decomposition generally refers to the disintegration of dead materials into particulate form or detritus, and the further breakdown of large particles to smaller and smaller particles, until the structure can no longer be recognized and complex organic molecules have been broken down to CO_2, H_2O and mineral components (Mason, 1997).

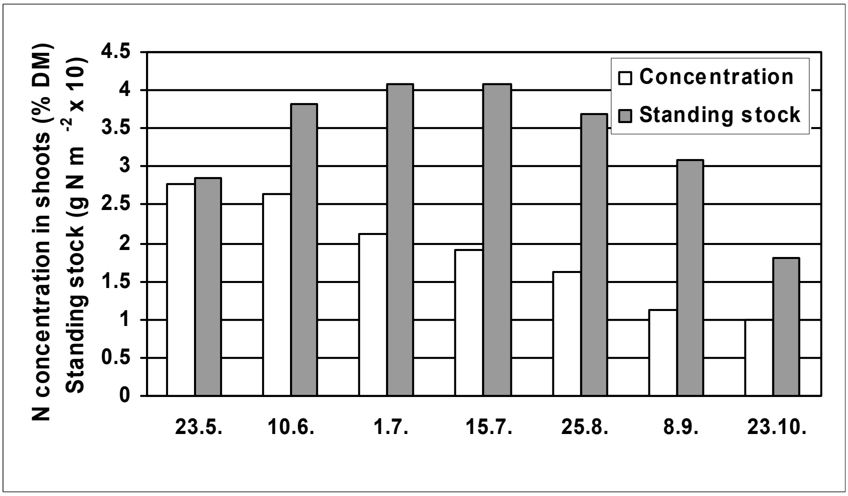


Figure 2-10. Concentration of nitrogen in aboveground shoots and respective nitrogen standing stocks in *Phragmites australis* from littoral of the Rožmberk Pond in south Bohemia during the growing season. Data from Dykyjová and Hradecká (1976).

The potential rate of nutrient uptake by the plant is limited by its net productivity (growth rate) and the concentration of nutrients in the plant tissue. Nutrient storage is similarly dependent on plant tissue nutrient concentrations, and also on the ultimate potential for biomass accumulation: that is, the maximum standing crop. Therefore, desirable traits of a plant used for nutrient assimilation and storage would include rapid growth, high tissue nutrient content, and the capability to attain a high standing crop (biomass per unit area) (Reddy and DeBusk, 1987).

In the literature, there are many reviews on nitrogen concentrations in plant tissue as well as nitrogen standing stocks for plants found in natural stands. Vymazal et al. (1999) in their survey on natural stands reported following ranges for *Phragmites australis* and *Phalaris arundinacea*: leaves 0.31 to 5.10% DM, stems 0.05 to 3.33% DM and shoots (i.e. the whole aboveground part) 0.78 to 3.9% DM. Johnston (1991) reported the range for nitrogen concentration in aboveground portions of herbaceous wetland vegetation at peak standing crop in the range of 0.5 to 3.94% DM with an arithmetic mean of 1.94% DM. Kadlec and Knight (1996) in their literature survey reported nitrogen concentration in plants typically used in constructed wetlands in the range of 1.46 to 3.95% DM with an average of 2.26% DM.

Reddy and DeBusk (1987), based on a literature survey, suggested the N standing stock was between 14 and 156 g N m^{-2} for *Typha, Phragmites,* *Scirpus* and *Juncus*. However, the values included also belowground standing stock which is generally not available for harvest and the authors indicated that usually >50% of the stock is stored in belowground biomass. Brix and Schierup (1989c) suggested that total nitrogen standing stock in emergent macrophytes is roughly between 20 and 250 g N m^{-2}. Vymazal (1995a) reported aboveground N standing stock in the range of 22 to 88 g N m^{-2} for 29 various emergent species. Johnston (1991) gives the range for nitrogen standing stock in emergent species between 0.6 and 72 g N m^{-2} with an arithmetic mean of 20.7 g N m^{-2}. Mitsch and Gosselink (2000) reported that the aboveground stock of nitrogen in freshwater marsh plants ranges from as low as 3 to 29 g N m^{-2}. Vymazal et al. (1999) reported nitrogen standing stock in aboveground biomass of *P. australis* and *P. arundinacea* growing in natural stands in the range of 0.04 to 63.4 g N m^{-2} and 2.0 to 15.5 g N m^{-2}, respectively while in constructed wetlands the respective values for both plants were 8.5 to 84 g N m^{-2} and 3.7 to 46.7 g N m^{-2}.

2.3.9 Ammonia adsorption

Ionized ammonia may be adsorbed from solution through a cation exchange reaction with detritus, inorganic sediments or soils. As a cation, it may be adsorbed by ion exchange mechanisms to colloidal mineral and organic components of the soil solid phase. Exchangeable ammonium is in equilibrium with the solution phase and may subsequently be released in a soluble form. Soluble and easily exchangeable ammonium are readily available for plant uptake or microbial immobilization. Ammonium is relatively immobile in soils and sediments. However, the low assimilatory demand of obligate anaerobes may result in considerable quantities, which exceed the capacity for effective retention by cation exchange processes. This may contribute to the high levels of ammonium in interstitial waters relative to levels of usually found in an oxidized surface water column (Graetz et al., 1973; Nissenbaum et al., 1972).

The adsorbed ammonia is bound loosely to the substrate and can be released easily when water chemistry conditions change. At a given ammonia concentration in the water column, a fixed amount of ammonia is adsorbed to and saturates the available attachment sites. When the ammonia concentration in the water column is reduced (e.g., as a result of nitrification), some ammonia will be desorbed to regain the equilibrium with the new concentration. If the ammonia concentration in the water column is increased, the adsorbed ammonia also will increase. If the wetland substrate is exposed to oxygen, perhaps by periodic draining, sorbed ammonium may be oxidized to nitrate (Kadlec and Knight, 1996). The Freundlich equation can be used to model ammonia sorption on the substrate (Sikora et al., 1995).

Ammonium ion (NH_4^+) is generally adsorbed as an exchangeable ion onto clays, and chemosorbed by humic substances, or fixed within the clay lattice. It appears that these reactions may occur simultaneously. The rate and extent of these reactions are reported to be influenced by several factors, such as nature and amount of clays, alternate submergence and drying, nature and amount of soil organic matter, period of submergence, presence of vegetation (Savant and De Datta, 1982).

Ammonium can be fixed in a plant unavailable form in the interlayer space of 2:1 type clays. The NH_4^+ ion occupies space usually filled with K^+, and the NH_4^+ is rendered unavailable when the layered clays contract, usually as a result of drying. Because NH_4^+ competes with K^+ for the fixation sites, fixation is usually low in soils with high available K^+ (Patrick et al., 1985).

2.3.10 Organic nitrogen burial

Some fractions of the organic nitrogen incorporated in detritus in a wetland may eventually become unavailable for additional nutrient cycling through the process of peat formation and burial. Johnston (1991) summarized that the nitrogen concentration in wetland soils (as dry mass) is in the range of 0.02 to 65 mg N g^{-1}; organic soils (17.1 mg g^{-1}) average about twice as much as mineral soils (8.3 mg g^{-1}). In natural, rich peatlands, the nitrogen content of these accumulated carbon reserves is typically about 25 to 30 mg N g^{-1} (i.e., 2.5 to 3.0% on a dry mass basis) (Richardson et al., 1978, Reddy et al., 1991). Under the more eutrophic conditions, the nitrogen content exceeds the top of this range (Kadlec and Knight, 1996). Typical peat formation rate in natural peatlands and bogs ranges from 10 to 180 g dry mass m^{-2} yr^{-1} (Richardson, 1989) but under more eutrophic conditions accretion is much larger (Kadlec and Knight, 1996). For example, in a lightly fertilized zone of Water Conservation Area 2A in the Florida Everglades reported values range from 460 g m^{-2} yr^{-1} (Craft and Richardson, 1993a) to 1130 g m^{-2} yr^{-1} (Reddy et al., 1991).

Johnston (1991) also reported that annual accumulation in wetland soils is in the range of 0.9 to 52.4 g N m^{-2} yr^{-1} with the average values of 1.6 and 14.6 g N m^{-2} yr^{-1} for organic and mineral soils, respectively. Nichols (1983) reported the nitrogen accumulation in organic wetland soils in the range of 0.1 to 4.7 and possibly up to 10 g N m^{-2} yr^{-1} for moderate to cold climates and warm, highly productive areas, respectively. Craft and Richardson (1998) reported the range of nitrogen accumulation between 1.2 and 16.6 g N m^{-2} yr^{-1} for organic soil freshwater wetlands in the United States with higher values for eutrophic sites.

The major N pools in natural wetlands are total sediment N (mostly organic N), total plant N, and available inorganic N in sediments (Bowden, 1987). The total sediment pool is the largest ranging from 100 to 1000 g N

m^{-2} in the upper 20 to 50 cm of wetland sediments. The total plant pool is roughly an order of magnitude less than total sediment N, while inorganic N pool in water is another order of magnitude less than the plant pool (Buresh et al., 1980b; Faulkner and Richardson, 1989; Richardson 1991; Kadlec and Knight, 1996). Verhoeven (1986), reviewing literature data, from Africa, America and Europe, reported that peat/litter compartment stores more than 97% of the nitrogen found in the upper 40-cm layer of peat mires.

2.4 Phosphorus transformations

The soil phosphorus cycle is fundamentally different from the N cycle. There are no valency changes during biotic assimilation of inorganic P or during decomposition of organic P by microorganisms. Soil P primarily occurs in the +5 (oxidized) valency state, because all lower oxidation states are thermodynamically unstable and readily oxidize to PO_4^{3-} even in highly reduced wetland soils (Lindsay, 1979). Phosphorus has only a minor gaseous phase (phosphine, PH_3) (Dévai et al., 1988, 1999; Dévai and DeLaune, 1995a; Roels et al., 2004; Niu et al., 2004). Phosphine is soluble in water, but has a high vapor pressure. It may be emitted from regions of extremely low redox potential, together with methane.

Phosphorus exists mainly as apatites, with a basic formula $M_{10}(PO_4)_6X_2$. Commonly the mineral (M) is calcium, less often Al or Fe. The anion (X) is either F^-, Cl^-, OH^- or CO_3^{2-}, thus there exist fluor-, chloro-, hydroxy- and carbonate apatites. Diverse substitutions and combinations of (M) and (X) result in some 200 forms of P occurring in nature (Paul and Clark, 1996).

Phosphorus does not show extensive biologically-induced fluxes to and from the atmosphere, as do carbon and nitrogen. Nor, in contrast to reduced forms of C and N, does it serve as a primary energy source for microbial oxidation. Nevertheless, soil organisms are intimately involved in the cycling of soil P. They participate in the solubilization of inorganic P and in the mineralization of organic P.

2.4.1 Forms of phosphorus in wetlands

Phosphorus in wetlands occurs as phosphate in organic and inorganic compounds. Orthophosphate occurs in ionic equilibrium, i.e., as:

$$\underset{}{H_3PO_4} \;\underset{pK = 2.2}{\Leftrightarrow}\; H_2PO_4^- \;\underset{pK = 7.2}{\Leftrightarrow}\; HPO_4^{2-} \;\underset{pK = 12.3}{\Leftrightarrow}\; PO_4^{3-} \qquad (2.32)$$

with $H_2PO_4^-$ and HPO_4^{2-} being the predominant species over pH range of 5 to 9 (Stumm and Morgan, 1970). Another group of inorganic phosphorus compounds are polyphosphates linearly condensed and cyclic. Organically-bound phosphorus is present e.g., in phospholipids, nucleic acids, nucleoproteins, phosphorylated sugars or organic condensed polyphosphates (coenzymes, ATP, ADP) (Syers et al., 1973; Vymazal, 1995a). Organic P forms can be generally grouped into 1) easily decomposable P (nucleic acids, phospholipids or sugar phosphates) and 2) slowly decomposable organic P (inositol phosphates or phytin) (Dunne and Reddy, 2005).

2.4.2 Phosphorus transformation and retention in wetlands

Phosphorus transformations in soil and water column of wetlands are manifold and include: adsorption and desorption, precipitation and dissolution, plant and microbial uptake, fragmentation and leaching, mineralization, sedimentation and burial (Fig. 2-11).

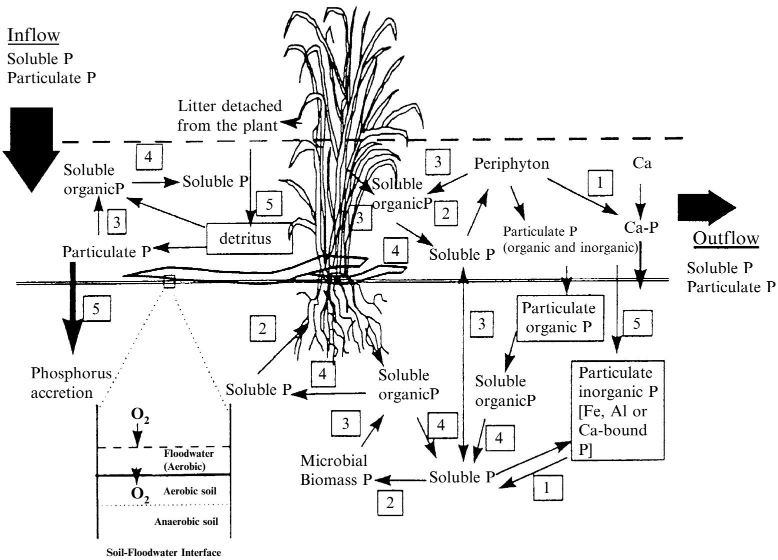


Figure 2-11. Phosphorus transformations in soil and water column of wetlands. 1) adsorption and desorption; precipitation and dissolution, 2) plant and microbial uptake, 3) fragmentation and leaching, 4) mineralization, 5) sedimentation and burial. From Reddy and D'Angelo (1996). With permission from Universität für Bodenkultur, Vienna, Austria.

Phosphorus that enters the wetland water column is rapidly absorbed by bacteria, periphyton, and plants. Radioisotope P studies have shown that 10 to 20% of the P is controlled by the biotic uptake initially (Davis, 1982; Richardson, 1985). Inorganic phosphorus transformations, subsequent complexes, and P retention in wetland soils and sediments are controlled by the interaction of redox potential, pH values, Fe, Al, and Ca minerals, organo-metallic complexes, organic matter content, clay minerals, hydraulic loading, and the amount of native soil P (Richardson, 1985, Faulkner and Richardson, 1989; Vymazal 1995a; Dunne and Reddy, 2005).

A large amounts of the phosphorus in the soil and sediments are present in the organic fraction. This complicates the system due to the dynamic equilibrium between organic and inorganic phosphorus forms. The reduction of redox potential following flooding can cause the transformation of crystalline Al and Fe minerals to the amorphous form. Amorphous Al and Fe hydrous oxides have higher P sorption capacity than crystalline oxides due to their larger number of singly coordinated surface hydroxyl groups (Williams et al., 1971; Richardson, 1999). Research by Petrovic and Kastelan-Macan (1996) suggests that the binding of inorganic P can involve the formation of a complex association of Ca, Mg, Fe and Al (pH dependent) bound to humic substances. Humic substances can act as bridges between humic macromolecules and phosphate ions. The other possibility is ligand exchange of the metal complex and the formation of insoluble metal-phosphates. In addition, Petrovic and Kastelan-Macan (1996) have shown in desorption experiments, high concentrations of fulvic acids compete for sorption sites on mineral particles. Such fulvics can increase P desorption by 10 to 20% (Kastelan-Macan and Petrovic, 1996). These findings have important consequences for storage and release of P in peat-based wetlands that have increased P additions (Richardson, 1999).

Phosphorus retention in wetlands can be defined as the capacity of the system to remove water column P through physical, chemical and biological processes, and retain it in a form that is not easily released under normal environmental conditions.

A theoretical hierarchy of the processes that control P retention in wetlands based on the literature survey is as follows (Reddy and Smith, 1987; Richardson and Craft, 1993; Richardson et al., 1997; Richardson, 1999): peat/soil accretion (high magnitude, very slow rate), soil adsorption (low to moderate magnitude, moderate rate), precipitation (moderate magnitude, fast rate), plant uptake (low to moderate magnitude, slow rate), detritus sorption (low magnitude, fast rate) and microbial uptake (very low magnitude, very fast rate).

Richardson and Marshall (1986) found that soil adsorption and peat accumulation (i.e., phosphorus stored in organic matter) control long-term phosphorus sequestration. But microorganisms and small sediments control initial uptake rates, especially during periods of low nutrient concentration and standing surface water. Both biotic and abiotic control mechanisms are thus functional in the peatland, and the proportional effect of each on P transfers is dependent on water levels, the amount of available P, fluctuating microorganism populations, seasonal changes in P absorption by macrophytes, and P soil adsorption capacity. Reddy et al. (1995) pointed out that long-term P retention is primarily controlled by the characteristics of bottom sediments and the P concentration in the water column. Productivity of aquatic vegetation is influenced by the P release from bottom sediments. Although aquatic vegetation and periphyton provide short-term retention and facilitate long-term P storage through accrual of organic matter, the overall long-term retention is determined by sediment and site characteristics (Hill, 1982; McCallister and Logan, 1978; Reddy et al., 1996).

2.4.2.1 Peat/soil accretion

Most studies on phosphorus cycling in wetlands have shown that soil/peat accumulation is the major long-term phosphorus sink and that natural wetlands are not particularly effective as phosphorus sink when compared with terrestrial ecosystems (Richardson, 1985). The sediment-litter compartment is the major P pool (>95%) in natural wetlands, with much lower plant pool and little in the overlying water (Verhoeven, 1986; Richardson and Marshall, 1986). Most soil P (>95%) in peatland systems is in the organic form (Verhoeven, 1986; Richardson et al., 1978; Richardson and Marshall, 1986) with cycling between pools controlled by biological forces (i.e., microbes and plants). The percentage of organic P is generally lower in wetlands with mineral substances (Faulkner and Richardson, 1989).

The peat accumulation rate in peatlands is very slow, with the world average of accretion being only 1 to 2 mm per year (Richardson and Nichols, 1985; Craft and Richardson, 1993a). Concomitant with peat building is P storage with values between 0.005 and 0.024 g m^{-2} yr^{-1} for unfertilized wetlands (Richardson, 1985). In nutrient-enriched wetlands long-term P accretion can reach nearly 1 g m^{-2} yr^{-1} (Craft and Richardson, 1993b). Craft and Richardson (1998) reported P accumulation rates in organic soil of freshwater wetlands in the United States in the range of 0.06 to 0.90 g m^{-2} yr^{-1}. Mitsch et al. (1979) reported annual P storage in an alluvial swamp in Illinois reached nearly 3.6 g m^{-2} yr^{-1} because of massive inputs of sediment during flooding conditions. Research to date would suggest that permanent storage of phosphorus in wetlands is below 1 g m^{-2} yr^{-1} and usually averages around 0.5 g m^{-2} yr^{-1} (Nichols, 1983; Richardson, 1985; Richardson and Marshall, 1986; Craft and Richardson, 1993b; Johnston, 1991). Therefore, Richardson and Qian (1999) found a clear pattern of low phosphorus output concentration when the total phosphorus mass loading rate of freshwater wetlands was less than 1 g m^{-2} yr^{-1}, but some variations among systems exist. The retention of phosphorus in constructed wetlands for wastewater treatment is usually much higher, up to about 150 g m^{-2} yr^{-1} but it is important to realize that P loading of natural wetlands is several orders lower as compared to treatment wetlands (Vymazal 1999a, 2005b; Vymazal et al., 2006a). For example, Richardson (1989) presented results from 11 natural wetland ecosystems from North America and Europe where inflow P loadings varied between 0.01 and 0.47 g m^{-2} yr^{-1} while Vymazal (1999a) reported that the inflow P loadings of 41 constructed wetlands from the Czech Republic, Poland and USA vary between 28.1 and 307 g m^{-2} yr^{-1}.

Johnston (1991) summarizing the huge literature data, reported phosphorus concentration in wetland soils in the range of 0.001 to 7.0 mg P g^{-1}; average P concentrations per unit mass being comparable for both organic and mineral soils. Nichols (1983) reported the phosphorus accumulation in organic wetland soils in the range of 0.005 to 0.22 and possibly up to 0.5 g P m^{-2} yr^{-1} for moderate to cold climates and warm, highly productive areas, respectively. Nixon and Lee (1986) summarizing data from various regions of the United States, reported the range of 0.003 to 20 mg P g^{-1}, the upper value being reported for a wetland receiving sewage effluent (Boyt et al., 1977). This value, however, is extremely high as phosphorus concentrations in soils are usually < 1.0 mg P g^{-1} (Nixon and Lee, 1986). Johnston (1991) reported the average soil standing stock of 64 (16 - 179) g P m^{-2}.

2.4.2.2 Soil adsorption and precipitation

Adsorption refers to movement of soluble inorganic P from soil porewater to soil mineral surfaces, where it accumulates without penetrating the soil surface. Phosphorus adsorption capacity of a soil generally increases with clay content or mineral components of that soil (Rhue and Harris, 1999). Munns and Fox (1976) pointed out that soluble phosphate added to soil usually adsorbs rapidly; but the concentration of phosphate in solution continues to decline slowly over a period of months. The term "slow reaction" or "slow fixation" usefully refers to the process or set of processes which cause the slow decline. The term "adsorption" can be restricted to the rapid process which closely approaches steady state within 1-7 days (Fox and Kamprath, 1970; Chen et al., 1973). The balance between P adsorption and desorption maintains the equilibrium between solid phase and P in soil porewater. This phenomenon is defined as phosphate buffering capacity, which is analogous to pH buffering capacity of a soil (Barrow, 1983; Rhue and Harriss, 1999).

Phosphorus is adsorbed on soil or sediment particles if sufficient Al, Fe, Ca and Mg are present. Which ion is more active in sequestering P depends on the pH of the system and the amount of the ion present (Andersen, 1975; Holford and Patrick, 1979; Robarge and Corey, 1979; Richardson, 1999). In acidic soils, inorganic P is adsorbed on hydrous oxides of Fe and Al, and P may precipitate as insoluble Fe-phosphates and Al-phosphates while sorption by Ca and Mg compounds is the dominant transformation at pH's greater than 8.0 (Kuo and Lotse, 1972; Syers et al., 1973; Parfitt et al., 1975; Rajan, 1975; Ryden et al., 1977; Ryden and Syers, 1977; Robarge and Corey, 1979; Berkheiser et al., 1980; Boström et al., 1982; Logan, 1982; Sonzongi et al., 1982; Froelich, 1988; Sanyal and DeDatta, 1991; Jugsujinda et al., 1995). Reddy et al. (1999) pointed out that availability is greatest in soils and sediments in slightly acidic to neutral pH. Adsorption of phosphorus is greater in mineral vs. organic soils (Richardson 1985; Richardson et al., 1978).

Maximum sorption capacity of a soil can be determined using a Langmuir model (e.g., Olsen and Watanabe, 1957; Syers et al., 1973; Holford and Mattingly, 1976; Taylor and Ellis, 1978). Other models commonly used in soil chemistry include Freundlich and Tempkin models (Reddy et al., 1999; Rhue and Harris, 1999). Although adsorption isotherms by themselves do not indicate the mechanism involved they do illustrate the equilibrium relationship between the amounts of adsorbed and dissolved species under given conditions (Taylor and Ellis, 1978).

Precipitation can refer to the reaction of phosphate ions with metallic cations such as Fe, Al, Ca or Mg, forming amorphous or poorly crystalline solids. These reactions typically occur at high concentrations of either phosphate or the metalloid cations (Rhue and Harris, 1999). A variety of cations can precipitate phosphate under certain conditions. Some important mineral precipitates in the wetland environment are (Emsley, 1980; Reddy and D'Angelo, 1994; Dunne and Reddy, 2005): apatite $Ca_5(Cl,F)(PO_4)_3$, hydroxylapatite $Ca_5(PO_4)_3OH$, variscite $Al(PO_4).2H_2O$, strengite $Fe(PO_4).2H_2O$, vivianite $Fe_3(PO_4)_2 . 8H_2O$, wavellite $Al_3(OH)_3(PO_4)_2 .5H_2O$, whitlockite $Ca_3(PO_4)2$ and fluoroapatite $Ca_5(PO_4)_3F$. In addition to direct chemical reaction, phosphorus can co-precipitate with other minerals, such as ferric oxyhydroxide and the carbonate minerals, such as calcite (calcium carbonate), $CaCO_3$.

The chemical processes of soil adsorption and precipitation are considered more important than uptake by plants and detritus, although rates would vary considerably among wetlands (Richardson and Craft, 1993b). For example, the amount of P stored by each process summarized for the Houghton Lake peatland in Michigan demonstrates that P adsorption can reach nearly 50% of total annual storage (Richardson and Marshall, 1986). A general comparison of the amount of P adsorption among wetlands soil types demonstrates that wetland systems vary greatly in terms of the amount of P that can be removed by this process. The system with the highest adsorption capacity has been shown to have mineral soils, which contain the highest amount of Fe or Al (Richardson, 1985). The high P adsorption capacity in relation to high Ca content was reported, for example, for the Florida Everglades soil (Richardson and Vaithiyanathan, 1995).

It has been reported that there is a significant correlation between amorphous and poorly crystalline forms of Fe and Al, which are typically extracted with an ammonium oxalate extraction, and P retention (sorption) by mineral soils (Berkheiser et al., 1980; Khalid et al., 1977; Richardson, 1985; Walbridge et al., 1991; Walbridge and Struthers, 1993; Gale, et al., 1994; Darke et al., 1997; Reddy et al., 1998). The primary sorption mechanism is thought to be ligand exchange, by replacement of surface hydroxyls on soil Al and Fe oxyhydroxides (Schwertmann and Taylor, 1989; Brady, 1990). Both noncrystalline and crystalline forms of Al and Fe sorb P by the same mechanism, but when present in significant amounts, noncrystalline forms tend to dominate soil adsorption reactions because of their larger surface-to-volume rations (Parfitt, 1989; Walbridge et al., 1991). Al gel has been shown to adsorb 30-70 times more P than gibbsite, and Fe gel can adsorb 10 times more P that goethite or hematite (Mc Laughlin et al., 1981).

Dunne and Reddy (2005) pointed out that it is important to note that in terrestrial soils such as those in grassland pastures, Fe and Al are typically found in crystalline forms, whereas in wetland soils, these ions typically occur in amorphous forms which have greater surface areas for P sorption reactions to occur. Also, the presence of organic matter in wetland soils and sediments is important in P retention process (Gale et al., 1994; Reddy et al., 1998). Iron and aluminum that complex with organic matter may be responsible for such a relationship, suggesting an indirect positive effect of organic matter complexes on P retention (Syers et al., 1973; Zhou et al., 1997; Rhue and Harris, 1999). In contrast, organic matter in terrestrial soils typically occludes P retention mechanisms with negative correlations between P sorption and organic matter reported (Dubius and Becquer, 2001).

Assessment of desorption rates are also required to determine net storage via this mechanism (Richardson, 1999). One of the proposed mechanisms for the release of phosphorus from soils upon submergence is the reductive dissolution of Fe(III) and Mn(IV) phosphate minerals (Patrick et al., 1973). The difference in behavior of phosphate under aerobic and anaerobic conditions is attributed to the change brought about in ferric oxyhydroxide by soil reduction. However, under anaerobic conditions the P associated with Fe may be released from soil to water (Wildung et al., 1977; Hosomi et al., 1981; Furumai and Ohgaki, 1982), whereas P bound to Al is not affected by changes in redox conditions (Moore and Reddy, 1994).

Within overlying wetland waters and within soil porewater, the solubility of P is influenced by pH and redox potential (e.g., Mortimer 1941, 1942; Andersen, 1975; Holford and Patrick, 1979). In a pH range 5-8, solubility is low at Eh of about 300 mV, which can result in low P concentrations in soil solutions (Patrick, 1968; Patrick et al., 1973). However, as the Eh decreases from 300 mV to -250 mV, P solubility increases at all pH levels, resulting in high P concentrations in soil porewater (Ann et al., 2000). Phosphorus solubility is highest under low pH and low Eh conditions. Under acidic conditions, an increase in P solubility is primarily due to reduction of ferric phosphate (Dunne and Reddy, 2005):

$$FePO_4 + H^+ + e^- = Fe^{2+} + HPO_4^{2-} \tag{2.33}$$

Dunne and Reddy (2005) pointed out that in soils dominated by Fe minerals, reduction of the soluble ferrous oxyhydroxide compounds results in amorphous "gel-like" reduced ferrous compounds with larger surface area than crystalline oxidized forms. A reduced soil has many sorption sites, as a result of the reduction of insoluble ferric oxyhydroxide compounds to more soluble ferrous oxyhydroxide compounds (Patrick and Khalid, 1974). Even though reduction increased sorption sites, these sites have lower P bonding energies for phosphate that do the smaller number of sites available in aerobic soils such as terrestrial grassland soils. Thus, a reduced soil will adsorb a large amount of P with a low bonding energy, thus desorption potential is high, while an oxidized soil will adsorb less P, but hold it more tightly (Patrick and Khalid, 1974).

In sulfate-dominated anaerobic soils and sediments, production of H_2S (through biological reduction of SO_4^{2-}) and formation of ferrous sulfides may preclude P retention by ferrous iron, hence, P is released (Caraco et al., 1990). In Fe and Ca dominated systems, Moore and Reddy (1994) observed that Fe oxides likely control the behavior of inorganic P under aerobic conditions, while Ca-phosphate mineral precipitation governs the solubility under anaerobic conditions.

However, the investigations have shown that not only under anaerobic, but also under aerobic conditions essential quantities of phosphate can be released from the sediment only by diffusion-controlled exchange processes (Vymazal, 1995a). Although release under aerobic conditions is about 10 times less than under anaerobic conditions the significance of aerobic P release should not be minimized. High bacterial activity at high temperatures can cause microanaerobic conditions at the sediment surface and the phosphorus release to aerobic water, in such cases, might be explained by the same process as the anaerobic release. Phosphorus release from sediments requires that mechanisms, which transfer phosphorus to the pool of dissolved phosphorus in the pore water, occur simultaneously, or within a short space of time, with processes which can transport the released phosphorus to the overlying water. Important "mobilization" processes are desorption, dissolution, ligand exchange mechanisms, and enzymatic hydrolysis (Boström et al., 1982). These processes are affected by a number of environmental factors, of which redox potential, pH and temperature are most important. Essential transport mechanisms are diffusion, wind-induced turbulence, bioturbation, current and gas convection (Boström et al., 1982).

Photosynthesis and respiration can initiate significant changes in water column pH on a diurnal basis. These processes can increase pH to values as high as 10, depending on the buffering capacity of the water column, precipitate a significant portion of water column P as calcium phosphate if dissolved Ca is available (Reddy et al., 1999). However, Diaz et al. (1994) noted that about 75 to 90% of the P precipitated was solubilized when pH levels decreased to below 8 as a result of an increase in CO_2 levels.

Retention of P by precipitation will be significant in waters with high Ca^{2+} and alkalinity but insignificant in poorly buffered waters (Reddy et al., 1999). In high Ca^{2+} and mildly alkaline waters, House (1990) attributed only 6% of the overall P removal to co-precipitation, while the reminder was due to biological uptake.

2.4.2.3 Microbiota uptake

The enormous numbers of microorganisms that inhabit soil mostly live in a state of near starvation, having a survival strategy that depends on the maintenance of ATP levels and an adenylate energy charge a measure of the stored metabolic energy (Brookes et al., 1983). Phosphorus has a key role in this strategy, participating in most significant metabolic pathways, as well as being a structural component of many biochemicals including nucleic acids, co-enzymes, phosphoproteins and phospholipids. Consequently, both plants and soil microorganisms actively compete for P from meagre levels of orthophosphate maintained in soil solution by chemical processes of precipitation-solubilization and adsorption-desorption (Tate, 1984).

Microbial uptake is very fast, but the magnitude (amount stored) is very low. The uptake by microbiota (bacteria, fungi, algae, microinvertebrates) is rapid because these organisms grow and multiply at high rates. Radio-phosphorus studies on wetland microcosms (Richardson and Marshall, 1986) indicated that phosphorus uptake by microbiota occurs on a time scale less than 1 hour. However, more than 90% was released within next 6 hours. In this study, the authors found that about 12-13% of the P in a northern peat-based fen (Houghton Lake, Michigan) was estimated to be stored (short-term storage) in microbial biomass. As compared to other compartments, microbial storage (0.5 - 1.0 g m^{-2} yr^{-1}) was the smallest (algae: 1.0 g m^{-2} yr^{-1}, 12-25%; macrophytes: 1.0 – 2.5 g m^{-2} yr^{-1}, 25-30%; soil adsorption: 1.5 – 3.8 g m^{-2} yr^{-1}, 38-46%). Total short-term storage of 4.0 - 8.3 g m^{-2} yr^{-1} is considerably higher as compared to long-term storage capacity based on 5 years field estimates 0.92 ± 0.15 g m^{-2} yr^{-1}. It seems that the amount of microbial storage depends also on trophic status of the wetland. In less enriched sites the microbial uptake may store more phosphorus as compared to more eutrofied sites (Richardson et al., 1997).

Soil microorganisms participate in the solubilization of soil P. Measurements of precise amounts of soil P solubilized by soil organisms is complicated by the concomitant mineralization of organic P. Attempts were made to isolate specific bacteria responsible for solubilization and most frequently isolated were species of *Pseudomonas* and *Bacillus* (Paul and Clark, 1996). Although bacteria are generally considered decomposers that simply mineralize organic P, they have also been shown to regulate the P flux across the sediment-water interface (Kleeberg and Schlungbaum, 1993) and contribute to terminal P burial through production of refractory organic compounds (Gächter and Meyer, 1993). The many factors that control the population dynamics and activities of soil microorganisms also control the balance between mineralization and immobilization of P in soil. They include temperature, moisture, aeration and the soil reaction (Tate, 1984).

Of importance, and seldom recognized, is the amount of P that can be sequestered by the algal component of wetlands, especially in areas with open water. Vymazal (1995a) pointed out that the role of algae in wetlands is mostly neglected despite the fact that algae can significantly influence the nutrient cycling and retention in wetlands. In addition, attention is usually paid only to attached forms (periphyton) while the role of plankton in wetlands has been assessed only rarely. Algae and algal assemblages can affect phosphorus cycling either directly (uptake, release) or indirectly through photosynthesis-induced changes in water and soil/water interface parameters (pH, dissolved oxygen).

It has been long known that algal cells are capable of "luxury uptake" whereby surplus P is accumulated in excess of that required for immediate growth. Luxury uptake provides the alga with a supply of phosphorus when external levels might otherwise be limiting. The excess phosphorus is built into polyphosphates. It has been suggested, however, that the storage function of polyphosphates is only secondary to their role in regulating the concentration of free phosphate ions in the cell. The formation of polyphosphates is an important difference between phosphorus metabolism in vascular plants and algae (Vymazal, 1995a).

There is good evidence that P is the dominant element controlling C and N immobilization in biological systems (Paul and Clark, 1996). In a classical and much discussed paper, Redfield (1958) hypothesized that P controls the C, N and S cycles of marine systems. He noted that the oceanic C:N:P ratio paralleled that of the plankton and believed the following general relationship to occur:

$$106CO_2 + 16NO_3^- + 122H_2O + 18H^+ \rightarrow C_{106}H_{263}O_{110}N_{16}P + 138O_2 \quad (2.34)$$

Paul and Clark (1996) pointed out that the aquatic algae and soil bacteria have a similar C:N ratio (106:16 for algae, 31:5 for soil bacteria). Algae have lower C:P ratio than soil bacteria (106:1 for algae, 31:1 for soil bacteria) but fall within the range for C:P found in soil organic matter (119:1 for cultivated or fertilized soils).

2.4.2.4 Plant uptake

Free orthophosphate is the only form of phosphorus believed to be utilized directly by algae and macrophytes and thus represents a major link between organic and inorganic phosphorus cycling in wetlands (Vymazal, 1995a). Organically bound P constitutes 30 to 50% of the total P in most soils but it may range from as low as 5% to as high as 95% (Paul and Clark, 1996). In agricultural soils, 30 to 70% of all the phosphate is present in an organic form; in nutrient-poor grasslands and forest soils this may be as much as 80 to 95% (Häussling and Marschner, 1989; Macklon et al., 1994) or 99% in organic tundra soils (Kielland, 1994). In swamp sediments, organic P may account for up to 87% of total P (Hesse, 1962), in Everglades histosols it is 40 to 90% (D'Angelo and Reddy, 1994), and in lake sediments 10 to 70% of total P is in organic form (Sommers et al., 1972). The importance of organic P mineralization in providing plant-available P was well established long time ago (e.g., Grunes et al., 1955; Haas et al., 1961; Dormaar, 1972). Some plant species can use phytate (inositol phosphate, a major component of the organic P fraction in soil), RNA, and glycerophosphate, in addition to inorganic phosphate, due to activity of phosphatases in the soil (Adams and Pate, 1992). Production of phosphatases by the roots provides an additional source of phosphate; these enzymes hydrolyze organic phosphate-containing compounds releasing inorganic phosphate that is absorbed by roots (Kroehler and Linkins, 1991).

Some plants adapted to low-phosphorus soils excrete acidifying and chelating compounds (e.g., citric acid, malic acid, and piscidic acid). Acidification enhances the solubility of phosphate in basic soils. Chelating compounds bind cations and thus release phosphate from sparingly soluble inorganic substrates. Both processes enhance the diffusion gradient for phosphate between the bulk soil and the root surface (Lambers et al., 1998). Citric acid releases phosphate from calcium phosphate complexes, whereas piscidic acid (p-hydroxybenzyl tartaric acid), in combination with reducing phenolics, is more effective in releasing phosphate from iron phosphate complexes (Marschner, 1995).

The macrophytes obtain and use phosphorus much more slowly than bacteria. Most of the phosphorus is taken up by plant roots, absorption through leaves and shoots is restricted to submerged species but this amount is usually very low (Carignan and Kalff 1980; Denny, 1980; Barko et al., 1991, Rattray et al., 1991). Phosphorus uptake by macrophytes is usually highest during the beginning of the growing season (in most regions during the early spring), before maximum growth rate is attained (Boyd, 1969, 1971; Dykyjová, 1973; Garver et al., 1988). This early uptake and storage of nutrients provide a competitive edge for growth-limiting nutrients during the periods of maximum demand (Boyd and Vickers, 1971). Biomass increases, however, should not be counted as part of the long-term sustainable phosphorus removal capacity of wetlands (Kadlec and Knight, 1996). The ratio of nitrogen to phosphorus remains surprisingly constant (8 to 10:1) when plants receive nutrients in a ratio similar to that in their tissues (Ingestad and Ågren, 1988), regardless of whether nutrients or light limit plant growth. A 10:1 N:P ratio is also found in plants sampled in the field for nonvascular plants, aquatic macrophytes, and vascular plants (Garten, 1976). This is slightly lower than the Redfield ratio (see section 2.4.2.3) for balanced N:P nutrition in algae. When plants deviate from the 10:1 N:P ratio, this generally reflects a nutritional imbalance caused by reduced uptake of the growth-limiting nutrient (Koerselman and Meuleman, 1996; Verhoeven et al., 1996), which is sometimes combined with luxury consumption of nutrients that do not limit growth (Lambers et al., 1998).

Another important response to seasons is the translocation of nutrients within the plant. Prior to autumn senescence, the majority of important ions is translocated from shoot portions to the roots and rhizomes. These stored nutrients are used during early spring growth (Dykyjová and Květ, 1982; Garver et al., 1988). Phosphorus storage in vegetation can range from short- to long-term, depending on type of vegetation, litter decomposition rates, leaching of P from detrital tissue, and translocation of P from aboveground to belowground biomass. Phosphorus storage in aboveground biomass of emergent macrophytes is usually short-term, with a large amount of P being released during decomposition of litter. The aboveground parts of most macrophytes grow and decay on a cycle ranging from the annual growing season in northern climates to faster cycles in southern climates (Reddy et al., 1999).

Phosphorus is released back from the biomass to the wetland ecosystem after the plant decay. The decomposition of aboveground litter and resultant release of nutrients to water involve at least two processes. An initial loss of soluble materials is attributed to abiotic leaching. This process is quite rapid and accounts for the majority of mass reduction during the early stages of decomposition. The rapid initial release of nutrients by leaching has been demonstrated in many wetland plants - up to 50% of nutrients are lost by leaching alone during the first few days of decomposition. Released nutrients may be incorporated into the protoplasm of decomposer organisms where activities such as respiration and denitrification account for additional nutrient loses (Vymazal, 1995a). However, dead roots decompose underground, therefore adding refractory compounds to subsurface soils and leachates to the porewater in the root zone. Thus, the aboveground portions of macrophyte return P to the water, while belowground portions returns P to the soil (Reddy et al., 1999).

The concentration of phosphorus in the plant tissue varies among species and sites and also it varies during the season (see section 2.3.8). The phosphorus concentrations in aboveground tissues usually do not exceed those found in plants growing in natural stands. Kadlec and Knight (1996) in their literature survey reported phosphorus concentration in plants typically used in constructed wetlands in the range of 0.08 to 0.63% dry matter with an average of 0.25% DM. Vymazal (1995a) reported maximum P concentration in aboveground tissue of 53 emergent plant species to be 0.64% DM. Vymazal et al. (1999) in their survey on natural stands reported following ranges for *Phragmites australis* and *Phalaris arundinacea*: leaves 0.03 to 0.40% DM, stems 0.02 to 0.43% DM and shoots (i.e. the whole aboveground part) 0.09 to 0.41% DM. For plants growing in constructed wetlands Vymazal et al. (1999) reported the ranges of 0.10 to 0.27% DM for leaves, 0.10 to 0.24% DM for stems and 0.04 to 0.36% DM for shoots.

Reddy and DeBusk (1987), based on literature survey, suggested the P standing stock in both above- and belowground biomass between 1.4 and 37.5 g P m^{-2} for *Typha, Phragmites, Scirpus* and *Juncus*. Brix and Schierup (1989c) suggested that standing stock in aboveground biomass of emergent macrophytes, and thus available for harvesting, is roughly between 3 and 15 g P m^{-2} yr^{-1}. Vymazal (1995a) reported aboveground P standing stock in the range of 0.1 to 11 g P m^{-2} yr^{-1} for 29 various emergent species and Vymazal (2004a) reported P aboveground standing stock in the range of 0.2 to 10.5 g P m^{-2} for emergent plants growing in 30 constructed wetlands in Europe, North America and Australia.

2.5 Sulfur transformations

Chemically, sulfur is one of the most interesting but also one of the most difficult elements; geochemically it is abundant, and biochemically it is most important. The complexity of sulfur chemistry originates from the many oxidation states sulfur can assume, as well as from the tendency of sulfur in the zero oxidation state to catenate, forming chains and rings of an astonishing variety. In addition, more than 200 years of scientific research on sulfur and its compounds has resulted in a vast body of literature which cannot easily be searched for reliable information. Moreover, this literature is full of errors and contradictions because earlier workers, not having the methods available that are standard today, often made claims that have not always been confirmed subsequently (Steudel, 2000).

Sulfur, the 14th most abundant element in the Earth's crust, occurs in nine valence states ranging from +6 in sulfate to -2 in sulfides. The reduced states (-2 and -1) are found in metal and metalloid sulfides of which pyrite (FeS_2) is the most abundant. The intermediate valencies (0, +2, +4) have only a transitory existence in the geochemical cycle ands are usually reaction intermediates (e.g. elemental sulfur in sulfide oxidation). Sulfate (+6) is the dominant oxidized form and it is the second most abundant anion in seawater and in freshwaters. Carbon disulfide (CS_2), carbonyl sulfide (OCS), dimethyl sulfide ($(CH_3)_2S$) and hydrogen sulfide (H_2S) are trace sulfur gases emitted by microorganisms (e.g., Lewis and Papavizas, 1970; Francis et al., 1974; Banwart and Bremner, 1976; Farwell et al., 1979) and found in most atmospheric and surface natural water samples (Turner and Liss, 1985; Kim and Andreae, 1987; Dévai and DeLaune, 1995b; Watts, 2000). Most sulfur species play a role in aqueous systems in which redox reactions occur either as a result of microbiological activity or simply following the thermodynamics of the system in non-enzymatic reactions (Middelburg, 2000; Steudel, 2000).

The biochemical oxidations and reductions of sulfur compounds constitute the biological sulfur cycle which is schematically shown in Figure 2-12. While the assimilatory reduction of sulfate is very common in prokaryotes, plants and fungi, the dissimilatory pathways are restricted to eubacteria and archaebacteria (Brüser et al., 2000). In natural habitats the pathways of the above sulfur cycle are usually interconnected. Sulfur compounds oxidizing and reducing bacteria may form a synthropical community which Baas-Becking (1925) termed "sulfuretum". In most sulfureta a "large sulfur cycle" between sulfide and sulfate can be observed.

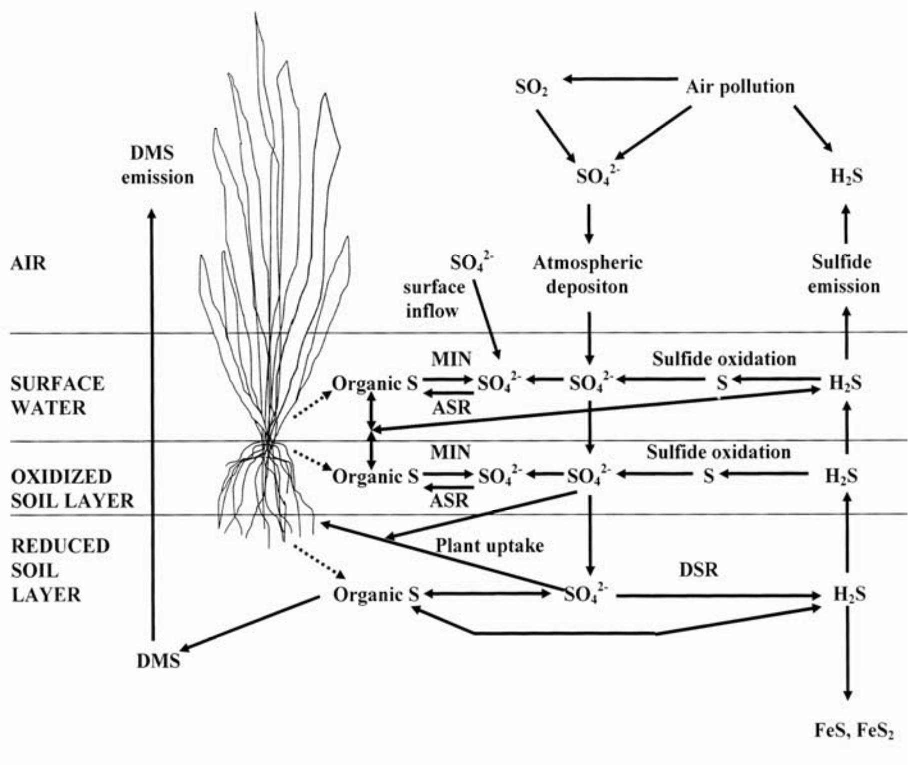


Figure 2-12. Transformations of sulfur in wetlands. ASR = assimilatory sulfate reduction, MIN = dissimilatory organic sulfur oxidation (mineralization), DSR = dissimilatory sulfate reduction, DMS = dimethyl sulfide. Modified from Mitsch and Gosselink (2000).

2.5.1 Assimilatory sulfate reduction

Most organisms are capable of using sulfate as a sole sulfur source for growth; they reduce it to hydrogen sulfide intracellularly and incorporate it into sulfur-containing amino acids such as cysteine which is then used as the building block for the other S-containing amino acids, which are then combined into proteins (Zehnder and Zinder, 1980; Killham, 1994; Paul and Clark, 1996; Brüser et al., 2000). In case there is no external reduced sulfur compounds available that can be incorporated into the organic matter of the organism, sulfate reduction to sulfide is required. Some bacteria are very much specialized for living in habitats with reduced sulfur compounds and such bacteria may lack a sulfate reduction pathway (e.g. Chlorobiaceae). Usually assimilatory sulfate reduction is a highly regulated process (Brüser et al., 2000). Assimilatory sulfate reduction is important in the formation of organic sulfur compounds and its main characteristics is that under normal conditions almost no sulfide is released from the cells (Zehnder and Zinder, 1980).

At least two assimilatory pathways are known by which sulfate can be reduced to sulfide. In one pathway APS (adenosine-5′-phosphosulfate), PAPS (3′-phosphoadenosine 5′-phosphosulfate) and sulfite are believed to be intermediates, the other pathway does not include the formation of PAPS and APS is directly reduced to sulfite. In both cases sulfate is actively transported into the cytoplasm (Sirko et al., 1995; Warthmann and Cypionka, 1996) and activated by an ATP sulfurylase (Brüser et al., 2000).

2.5.2 Dissimilatory sulfate reduction

Under reducing conditions, dissimilatory sulfate reduction transforms SO_4^{2-} to H_2S during respiration by several genera of strictly anaerobic bacteria by reaction with a variety of organic substrates (generically written as CH_2O), which can generate an alkalinity effect due to protons that are consumed during the reaction (Gambrell and Patrick, 1978; Zehnder and Zinder, 1980; Laanbroek and Veldkamp, 1982; Brown, 1985; Spratt et al., 1987; Mandernack et al., 2000):

$$2CH_2O + SO_4^{2-} + H^+ \rightarrow 2CO_2 + HS^- + 2H_2O \tag{2.35}$$

The sulfate reducing bacteria play a key role in the mineralization processes in marine sediments. Sulfate reduction in salt marshes and nearshore sediments is reported to account for 25 to 90% of total respiration (Skyring et al., 1978; Howarth, 1979, 1984; Howarth and Teal, 1980; Jørgensen, 1982; Howart and Giblin, 1983; Luther et al., 1986). Significant sulfate reduction rates have also been reported for fresh water sediments, despite low sulfate concentrations in the pore water (Ingvorsen et al., 1981; King and Klug, 1982; Hordijk et al., 1985; Dunnette, 1989; Hadas and Pinkas, 1992). This finding suggest that sulfate reducing bacteria in freshwater have acquired high affinity sulfate uptake systems to cope with the low sulfate concentrations (Smith and Klug, 1981; Ingvorsen and Jørgensen, 1984).

Laanbroek and Veldkamp (1982) in their review on sulfate reducing bacteria pointed out that after their discovery by Beijerinck (1895) the knowledge of this group of organisms accumulated only very slowly. Even in 1949, only few pure cultures were available (Butlin et al., 1949) and the organic substrates mainly used as carbon and energy sources, malate and lactate, were oxidized to the acetate level. It was not until 1930 that Baars described the acetate-oxidizing *Desulfovibrio rubentschickii*. A major breakthrough in the field of sulfate reduction came through the work of Widdel (1980) who described no less than six new species, belonging to five new genera.

The sulfate reducing bacteria have been traditionally considered to consist of a small group of highly specialized anaerobic bacteria with similar physiological and bioenergetic systems (Postagte, 1979). However, the evidence indicates that the sulfate-reducing bacteria encompass a much larger variety of bacteria than previously suspected (Pfenning and Widdel, 1981; Paul and Clark, 1996), and have number of growth modes other than sulfate reduction (Peck and LeGall, 1982). Sulfate-reducing bacteria use low molecular weight organic acids, alcohols, and , often H_2 as electron donors. These organisms are responsible for sulfide formation in waterlogged soils and sediments. Dissimilatory sulfate reduction is now recognized in a number of bacterial genera, e.g.,: *Desulfovibrio*, *Desulfotomaculum*, *Desulfomonas*, *Desulfococcus*, *Desulfobacter*, *Desulfobulbus*, *Desulfosarcina*, *Desulfonema*, *Desulfuromonas*, *Thermodesulfobacterium,* and others, whose name usually begins with “Desulfo-” (Paul and Clark, 1996; Widdel and Hansen, 1992). Different genera of sulfate-reducing bacteria can coexist at one location.

Two groups of sulfate-reducing bacteria can be distinguished: a group oxidizing organic compounds to the level of fatty acids (Eq. 2.36) and a group oxidizing organic matter completely to carbon dioxide (Eq. 2.37) (Laanbroek, 1990). Hydrogen sulfide as the end-product of the sulfate reduction is excreted by the cells as opposed to the assimilatory reaction (Zehnder and Zinder, 1980; see section 2.5.1).

$$SO_4^{2-} + \text{lactate} \rightarrow H_2S + \text{acetate} + CO_2 \tag{2.36}$$

$$SO_4^{2-} + \text{acetate} \rightarrow H_2S + CO_2 \tag{2.37}$$

During dissimilatory sulfate reduction, only a small percentage of the energy of the organic matter which is respired is available to the microbes for growth and maintenance. Most of the energy of the organic matter is transferred and stored as the reduced sulfur products of sulfate reduction (Howarth and Teal, 1980). Reduced sulfur compounds such as hydrogen sulfide then becomes important carriers of energy and can support bacterial autotrophic fixation of CO_2 either through photosynthesis in the presence of light or through chemolithotrophy in the presence of an oxidizer such as O_2 or nitrate (Cohen et al., 1975; Kuenen, 1975; Pfenning, 1975; Kelly, 1982; Kuenen and Beudeker, 1982; see also section 2.5.3). Sulfide has been shown to be a soil phytotoxin. It has been postulated to limit nutrient uptake and therefore growth in marshes (Mendelssohn and Seneca, 1980; Morris, 1980; Koch and Mendelssohn, 1989). Sulfide has also been found to inhibit respiratory metallo-enzyme activities (cytochrom oxidase, catalase, peroxidase, ascorbic acid oxidase and polyphenol oxidase) in wetland plants (Allam and Hollis, 1972). Sulfide also significantly reduces external phosphatase activity in marsh plant species (Havill et al., 1985).

Jakobsen et al. (1981) pointed out that the redox potential in soils and sediments at which sulfide formation occurs has variously been reported as +300 mV to +250 mV (Jørgensen, 1977b), +100 mV (Baas Becking et al., 1960), -75 mV (Harter and McLean, 1965), -150 mV (Connell and Patrick, 1968), and -200 mV (Postagte, 1959). The pH for sulfate reduction has been reported to be in the range of 4 to 10 (Baas Becking et al., 1960) while Connell and Patrick (1968) found that sulfate reduction occurred only in the pH range 6.5 to 8.5, with a sharp optimum at pH 6.8. These variations seem to indicate that sulfide formation is very dependent on the environmental conditions. De Laune et al. (2002) reported that sulfate reduction rates in freshwater wetlands range between 85 and 1,470 g S m^{-2} yr^{-1} while sulfate reduction in salt marshes is generally higher, up to 2,250 g S m^{-2} yr^{-1}.

Megonikal et al. (2004) in their excellent review on anaerobic metabolism reported that besides the common fermentation products, sulfate-reducing bacteria have been shown to consume a variety of other substrates including xenobiotics and other aromatic compounds (Bolliger et al., 2001; Elshahed and McInerney, 2001; Kniemeyer et al., 2003; Kuever et al., 2001; Lovley et al., 1995), carbohydrates such as fructose and sucrose (Sass et al., 2002), amino acids (Burdige, 1989; Coleman, 1960), alkanes and alkenes up to C20 (Aeckersberg et al., 1991, 1998), phosphite (Schink et al., 2002), aldehydes (Tasaki et al., 1992), dicarboxylic acids (Postgate, 1984), glycolate (Friedrich and Schink, 1995), methylated nitrogen and sulfur compounds (Finster et al., 1997; Heijthuijsen and Hansen, 1989; Kiene, 1988; van der Maarel et al., 1996), acetone (Platen et al., 1990), and sulfonates such as taurine and cysteate (Visscher et al., 1999).

Nitrate and nitrite reduction are rather widespread in sulfate reducers (Keith and Herbert, 1983; McCready et al., 1983; Mitchell et al., 1986; Moura et al., 1997; Seitz and Cypionka, 1986), and in some cases nitrate is preferred over sulfate (Seitz and Cypionka, 1986).

Sulfate is reduced to sulfide within the anaerobic zones and is stable while within the reduced environment. The free dissolved sulfide may react with metals, principally iron, and precipitate as ferrous sulfide. The firstly precipitated mineral, hydrotroilite ($FeS.nH_2O$) is labile under acidic conditions and is termed acid-volatile sulfide. After diagenesis the hydrotroilite gives pyrite (FeS_2) which is a stable mineral, not susceptible to autooxidation (Nedwell and Abram, 1978):

$$SO_4^{2-} \leftrightarrow S^{2-} + Fe^{2+} \leftrightarrow FeS.nH_2O \leftrightarrow FeS_2 \quad (2.38)$$

However, only a small fraction of the total annual sulfide yield may actually be precipitated within anaerobic sediments (Nedwell and Floodgate, 1972; Rickard, 1975; Howarth, 1979; Howarth and Giblin, 1983; Krairapanond et al., 1992). Sulfide is autoxidizable in the presence of of oxygen (Almgren and Hagstrom, 1974; Sorokin, 1972), and sulfide concentrations are therefore lowest at the surface of the sediment. This concentration gradient results in the diffusion of sulfide towards the aerobic sediment surface where it may be autoxidized, or alternately may be biologically oxidized or released to the atmosphere (Nedwell and Abram, 1978; Adams et al., 1979; see also Figure 2-12). The sulfide represents a free energy source under aerobic conditions, and aerobic chemotrophic bacteria such as the Thiobacillaceae or Beggiatoaceae may use the sulfide as an electron donor; while phototrophic bacteria (Chromatiaceae, Chlorobacteriaceae) are able to use the sulfide as a reductant for photosynthesis (Blackburn et al., 1975; see also section 2.5.3). However, in some marsh sediments pyrite forms quickly and is the major end product of sulfate reduction (Howarth, 1979; Howarth and Teal, 1979; Howarth and Giblin, 1983).

The several steps to the process of oxidizing pyrite to produce acid and other products involve chemical and microbiological processes. Reaction with dissolved oxygen is one of the two ways pyrite produces ferrous iron and sulfur (Prasittikhet and Gambrell, 1989):

$$FeS_2 + 1/2O_2 + 2H^+ \rightarrow Fe^{2+} + 2S + H_2O \quad (2.39)$$

Sulfate forms from oxidation of sulfur:

$$S + 3/2O_2 + H_2O \rightarrow SO_4^{2-} + 2\ H^+ \quad (2.40)$$

The above equations show that pyrite exposed to the atmosphere will oxidize chemically yielding Fe^{2+}, sulfate and sulfuric acid. The summary reaction is given below (Wainwright, 1984):

$$2FeS_2 + 7O_2 + 2H_2O \rightarrow 2FeSO_4 + 2H_2SO_4 \quad (2.41)$$

Complete oxidation and hydrolysis of the iron to ferric oxide yields two moles of sulfuric acid per mole of pyrite oxidized (van Breemen, 1982):

$$FeS_2 + 15/4O_2 + 7/2H_2O \rightarrow Fe(OH)_3 + 2SO_4^{2-} + 4H^+ \quad (2.42)$$

The above reaction is sulfuricization. At a near neutral soil pH, the process is relative slow, but it accelerates as acidity increases. In the presence of oxygen, the ferrous ion (Fe^{2+}) produced by these reactions is oxidized to ferric iron, normally a slow reaction at low pH (Singer and Stumm, 1970). However, *Thiobacillus ferrooxidans*, which functions well between pH 2.5 and 5.8 (Goldhaber and Kaplan, 1974), is effective in oxidizing reduced sulfur species and also ferrous iron at these low pH levels and thus returns ferric iron to the system (Wainwright, 1984):

$$4FeSO_4 + O_2 + 2H_2SO_4 \rightarrow 2Fe_2(SO_4)_3 + 2H_2O \quad (2.43)$$

When the pH of an oxidized system is sufficiently low for Fe^{3+} to exist in solution, Fe^{3+} may catalyze the oxidation of pyrite. The dissolved Fe^{3+} favors rapid oxidation of pyrite, especially as the pH of an oxidized system decreases below 4, according to the reaction below (Wainwright, 1984; Prasittikhet and Gambrell, 1989):

$$FeS_2 + Fe_2(SO_4)_3 \rightarrow 3\,FeSO_4 + 2S \quad (2.44)$$

$$2S + 6Fe_2(SO_4)_3 + 8H_2O \rightarrow 12FeSO_4 + 8H_2SO_4 \quad (2.45)$$

At high pH, the oxidation of pyrite by Fe^{3+} ions is limited because Fe^{3+} is appreciably soluble only at low pH less than 4, and *Thiobacillus ferrooxidans*, on the other hand, does not function at a high pH. Thus, in soils with high pH, ferric oxides and pyrite may be in close physical proximity, but the rate of pyrite oxidation by this process is limited by the insolubility of ferric ion (Prasittikhet and Gambrell, 1989).

2.5.3 Inorganic sulfur oxidation

In the presence of available electron acceptors, sulfide, elemental S, thiosulfate, and tetrathionate are oxidized by both chemical and biological pathways (Wainwright, 1984; Paul and Clark, 1996):

$$SH^- \rightarrow S^o \rightarrow S_2O_3^{2-} \rightarrow S_4O_6^{2-} \rightarrow SO_3^{2-} \rightarrow SO_4^{2-} \quad (2.46)$$

Sulfide and oxygen react chemically in aqueous systems but the presence of microorganisms can accelerate this process. Usually this kind of microbial sulfide oxidation is found in habitats in which sulfide and oxygen meet such as in waters overlying anaerobic sediments or anaerobic-aerobic interfaces in soil microenvironments. In these zones sulfide is oxidized by the chemosynthetic colorless sulfur bacteria which are mostly and are of two types (Zehnder and Zinder, 1980; Kuenen et al., 1985). The first type deposits sulfur inside the cell which accumulates as long as H_2S is available:

$$H_2S + 0.5O_2 \rightarrow S^o + H_2O \tag{2.47}$$

As sulfide sources are depleted, the internally stored sulfur is oxidized and sulfate is released (Pringsheim, 1967):

$$S^o + 1.5O_2 + H_2O \rightarrow H_2SO_4 \tag{2.48}$$

Beggiatoa, a long filamentous gliding bacterium, and *Thiothrix* are common bacteria that oxidize H_2S with deposition of sulfur intracellularly (Kowallik and Pringsheim, 1966; Shively, 1974; Leadbetter, 1974; Strohl and Larkin, 1978; Grant and Bathman, 1987). Colorless sulfur bacteria of the genus *Beggiatoa* are among the largest and most conspicuous of all the bacteria. In the nature, the filaments only seem to grow where both H_2S and O_2 are present (Jørgensen, 1977a; Kuenen and Beudeker, 1982). Since H_2S is not stable in oxic waters due to auto-catalytic oxidation by O_2, the habitat of *Beggiatoa* is restricted to the transition zone between oxic and anoxic environments where O_2 and H_2S are continuously supplied by diffusion along opposite gradients. Where these gradients are steep, *Beggiatoa* and other types of colorless sulfur bacteria may form white patches of dense cell masses (Jørgensen, 1977a; Whitcomb et al., 1989). Oxidation of sulfide to sulfate, via S^o intermediate, was described for *Beggiatoa* more than 100 years ago by Winogradsky (1887, 1888).

By similar reactions, a second type of chemosynthetic sulfur-oxidizing bacteria deposits sulfur outside of the cell. This assemblage is represented best by the genus *Thiobacillus*, which oxidizes sulfide, S^o, and other reduced sulfur compounds such as thiosulfate (Wetzel, 2001).

The other major group of sulfur-oxidizing bacteria is the photosynthetic (colored) sulfur bacteria, anaerobs that can be divided conveniently into the green sulfur bacteria (Chlorobacteriaceae) and purple sulfur bacteria (Thiorhodaceae) (Wetzel, 2001). Both forms are commonly found in mud and stagnant waters containing H_2S and exposed to light. They reoxidize H_2S, coming from lower anaerobic layers. They require light as an energy source and H_2S as an electron donor in the photosynthetic reduction of CO_2 (Trudinger, 1979; Paul and Clark, 1996; Wetzel, 2001):

$$CO_2 + 2H_2S \xrightarrow{\text{light}} (CH_2O) + H_2O + 2S \tag{2.49}$$

$$2CO_2 + 2H_2O + H_2S \xrightarrow{\text{light}} 2(CH_2O) + H_2SO_4 \tag{2.50}$$

$$3CO_2 + 5H_2O + 2S \xrightarrow{\text{light}} 3(CH_2O) + 2H_2SO_4 \tag{2.51}$$

Zehnder and Zinder (1980) pointed out that that elemental sulfur is often an intermediate, while the end product is H_2SO_4. The green sulfur bacteria have that color because of the presence of bacteriochlorophyll. They can be brown in color if carotenoids are present. These members of the family Chlorobiaceae have external S granules in the presence of HS^-. The purple sulfur bacteria are divided into two groups. The members of family Chromatiaceae store elemental sulfur internally in the presence of HS^-. *Chromatium*, *Thiospirillum* and *Thiocapsa* are the best know genera. Members of the family Ectothiorhodospiraceae store the elemental sulfur externally (Paul and Clark, 1996). More details on photosynthetic sulfur bacteria could be find, for example, in reviews by Pfennig (1967), Madigan (1988), Stolz (1991) or Friedrich (1998).

2.6 Iron and manganese

2.6.1 Iron

In nature, iron exists as common minerals, including those of carbonates (siderite, $FeCO_3$); oxides (hematite, Fe_2O_3; magnetite Fe_3O_4; limonite $2Fe_2O_3$. H2O), sulfides (troilite, FeS; pyrite, FeS_2); sulfates (melanterite, FeSO4 . 7H2O) and silicates (thuringite, $7FeO \cdot 3(Al,Fe)_2O_3 \cdot 5SiO_2 \cdot n\,H_2O$, chamosite, $4FeO \cdot Al_2O_3 \cdot 3\,SiO_2 \cdot 3H_2O$). Iron is frequently associated with organic material in nature, and is perhaps the most versatile of the biocatalytic elements. Its many chemical reactivities are due to its two valence states, Fe^{2+} and Fe^{3+} as well as range of oxidation-reduction potentials associated with the $Fe^{2+} \leftrightarrow Fe^{3+}$ transition in various iron-containing compounds (Lundgren and Dean, 1979).

2.6.1.1 Iron oxidation (deposition)

At neutral pH and positive redox potentials, oxidized iron is relatively stable (Patrick and Henderson, 1981). Hence, the oxidation of ferrous ions may proceed almost completely chemically under these conditions and it is better to speak about iron-depositing bacteria than of iron-oxidizing bacteria (Jones, 1986). Direct involvement of bacteria in the oxidation of ferrous ions is likely at low pH (Laanbroek, 1990; Lovley, 1995). Several genera of bacteria seem to be involved in the deposition of ferric ions (Van Veen et al., 1978; Ghiorse, 1984; Jones, 1986) among them filamentous (such as *Sphaerotilus* and *Leptothrix*), prosthecate (e.g., *Gallionella*) and encapsulated and coccoid bacteria (*Metallogenium* and *Siderocapsa*). Fe(III) is generally deposited on outer cell surface that are usually anionic in charge.

Iron deposition bacteria develop at the sites of a redox gradient, usually dominated by the Fe(II)/Fe(III) couple. The redox gradients can be as large as several meters of water strata in the metalimnion and upper hypolimnion of eutrophic lakes or only a few centimeters at oxic/anoxic interfaces. Chemosynthetic utilization of energy from inorganic oxidation is relatively inefficient, especially in the case of oxidation of iron (and manganese) (Wetzel, 2001).

The oxidation of Fe(II) to Fe(III) and subsequent hydrolysis to ferric hydroxide, $Fe(OH)_3$ or oxy-hydroxide, FeOOH (Eqs. 2.52., 2.53 and 2.54, Hall et al., 2005) proceeds spontaneously in the presence of oxygen but this reaction is strongly dependent upon the redox potential and pH (Singer and Stumm, 1970). Stumm and Morgan (1970) reported that under aerobic conditions, the oxidation of Fe(II) is very slow below pH 6 but the activity diagram presented by Singer and Stumm (1970) showed that substantial reduction in the rate of abiotic oxidation of Fe(II) only occurs below pH 5.

$$Fe^{2+} + 0.25O_2 + H^+ \rightarrow Fe^{3+} + 0.5H_2O \tag{2.52}$$

$$3H_2O + Fe^{3+} \rightarrow Fe(OH)_3 + 3H^+ \tag{2.53}$$

$$2H_2O + Fe^{3+} \rightarrow FeOOH + 3H^+ \tag{2.54}$$

The true iron bacteria occur in iron-rich waters of neutral or alkaline pH. Characteristic reactions of the few chemoautotrophic bacteria that deposit hydroxides and oxides are (Wetzel, 2001):

$$4Fe(HCO_3)_2 + O_2 + 6H_2O \rightarrow 4Fe(OH)_3 + 4H_2CO_3 + 4CO_2 \tag{2.55}$$

Under acidic conditions, several autotrophic bacteria (e.g., *Thiobacillus ferrooxidans*, *T. thiooxidans*, *Sulfolobus acidocladarius*) can oxidize iron disulfides, and sulfur (S^o) to sulfate, with Fe(III) serving as the electron acceptor (Eqs. 2.44 and 2.45). Ferric sulfate, the product of the oxidation, reacts with water to form ferric hydroxide and sulfuric acid (Eq. 2.56). This reaction is spontaneous and leads to a net increase in acid formation in the environment (Lundgren and Dean, 1979):

$$2Fe_2(SO_4)_3 + 12H_2O \rightarrow 4Fe(OH)_3 + 6\ H_2SO_4 \tag{2.56}$$

Jacob and Otte (2003) pointed out that the precipitation of iron (oxy-) hydroxides in the rhizosphere, due to radial oxygen loss (ROL) and the action of microorganisms, has been well described and is known to potentially lead to the formation of so-called iron plaque (Chen et al., 1980; Mendelssohn and Postek, 1982; Taylor et al., 1984; McLaughlin et al., 1985; Conlin and Crowder, 1989; Otte et al., 1989; St.Cyr and Crowder, 1990;

Crowder and St-Cyr, 1991; Ye et al., 1997, 1998). Macfie and Crowder (1987) pointed out that in natural wetlands all of the factors which affect the solubility of iron in the soil may determine the presence and extent of plaque accumulation on plat roots. These factors may include pH (Sarkar and Wyn Jones, 1982; Vose, 1982; Tiller et al., 1984; Uren, 1984; Bjerre and Schierup, 1985), inorganic carbonates (Boxma, 1972; Mengel et al., 1984), other soluble salts (Loeppert et al., 1984), soil organic matter content (Schelske, 1962; Bolter and Butz, 1977; Tiller et al., 1984), cation exchange capacity (Barber, 1984), the total amount of iron in the soil (Bjerre and Schierup, 1985), soil particle size (Smeulders et al., 1983), the presence of natural chelating agents such as those released from decaying organic matter or products of microbial activity (Elgala and Amberger, 1982; Sarkar and Wyn Jones, 1982), and other metals which may compete for adsorption sites (Bjerre and Schierup, 1985). It has been suggested that iron plaque may act as a barrier to toxic substances uptake, mainly heavy metals, owing to adsorption and immobilization of metals by iron plaque (Taylor and Crowder, 1983; Taylor et al., 1984; Otte et al., 1987; Greipson and Crowder, 1992; Noller et al., 1994).

Anaerobic Fe(II) oxidation is one of the most recently recognized categories of anaerobic metabolism. The first report of such metabolism described anaerobic, phototrophic organisms that required only Fe(II), CO_2 and light for growth (Widdel et al., 1993). Since then, several such bacteria have been isolated from freshwater and marine environments (Ehrenreich and Widdel, 1994; Heising and Schink, 1998; Heising et al., 1999; Straub et al., 1999). Little is known about the physiology or ecology of these organisms (Straub et al., 2001) but most strains can couple Fe(II) oxidation to NO3- reduction (Straub et al., 1996; Hafenbradl et al., 1996).

2.6.1.2 Iron reduction

Reduction of Fe(III) in anaerobic aquatic environments may involve both microbial dissimilatory reduction and chemical reduction (Froelich et al., 1979; Berner, 1980; Sørensen, 1982; Hines et al., 1984; Aller et al., 1986; Jacobson et al., 1987; Lovley, 1991; Jacobson, 1994). Under natural conditions, the reduction of ferric oxide may proceed chemically (Eqs. 2.57-2.58) by the involvement of sulfide or organic materials such as phenols and carboxylic acid (Aller et al., 1986; Lovley, 1987). So, sulfate-reducing, sulfite producing bacteria may be indirectly involved in the reduction of ferric oxide (Laanbroek, 1990). When sulfide oxidation occurs, in addition to Fe(III) reduction, the same overall reaction will produce Fe(II) without any apparent sulfate reduction (Eqs. 2.57, 2.58):

$$2FeOOH + 4H^+ + H_2S \leftrightarrow 2Fe^{2+} + S^o + 4H_2O \tag{2.57}$$

$$6FeOOH + 10H^+ + S^o \leftrightarrow 6Fe^{2+} + SO_4^{2-} + 8H_2O \tag{2.58}$$

When the reduction of nitrate stops by depletion of this electron acceptor, the reduction of ferric oxide starts. Ferric oxides are assumed to be one of the most abundant electron acceptors in soils as well as in sediments (Ponnamperuma, 1972; Gotoh and Patrick, 1974; Lovley and Phillips, 1986a).

A wide range of anaerobic bacteria are able to conserve energy through the reduction of Fe^{3+} to Fe^{2+} (Laanbroek, 1990; Lovley, 1991; Lovley et al., 1993; Vymazal 1995a; Vargas et al., 1998; Bridge and Johnson, 1998, 2000; Coates et al., 2001; Johnson and Hallberg, 2005). Most Fe^{3+}-reducing organisms have the capacity to reduce Mn^{4+} to Mn^{2+} and at least one other common oxidant such as O_2, NO_3^-, SO_4^{2-}, elemental sulfur (S^o), or humic substances (Thamdrup, 2000; Senko and Stolz, 2001). Most of the known species of Fe(III)-respiring bacteria are in the delta subclass of the *Proteobacteria*. This subclass includes the Geobacteraceae family, which is populated entirely by Fe(III)-reducing bacteria, including the well-studied species *Geobacter metallireductans* (Lovley et al., 1993).

Iron and manganese-reducing microorganisms can dissolve insoluble Fe^{3+} and Mn^{4+} oxides, resulting in the release of soluble Fe^{2+} and Mn^{2+}. At the same time, other metals which were bound in oxidized forms, could be released as well. These microorganisms can thus affect the fate of other contaminant metals through both direct enzymatic reduction and indirect reduction catalyzed by biogenic Fe^{2+} and Mn^{2+}. The bacterial reductive dissolution of ferric iron minerals follows the equation and may represent a significant pathway for organic matter mineralization (Kosolapov et al., 2004):

$$4\,Fe(OH)_3 + CH_2O \rightarrow 4\,Fe^{2+} + HCO_3^- + 7\,OH^- + 3\,H_2O \qquad (2.59)$$

Two types of ferric oxide-reducing bacteria can be distinguished: a group producing fermentation products and a group using fermentation products for the generation of energy (Lovley, 1987; Lovley and Phillips, 1989). Facultative and strictly anaerobic bacteria belong to the first group, many of which are also able to reduce nitrate. Ferric oxide does not seem to be reduced by the enzyme nitrate reductase in those bacteria, but by separate ferric oxide reductases (Laanbroek, 1990).

Different forms of ferric oxidex, often in complex mixtures with each other, that range widely in degree of crystallinity, particle size, available surface area reactivity and oxidation state, exist in aerobic drained as well as in waterlogged soils (Gotoh and Patrick, 1974; Murray, 1979; Kalhon and Emerson, 1984; Murray et al., 1984). The Fe(III) and Mn(IV) oxides are often considered to exist as coatings on clays and other particles (Carroll, 1958; Patrick and Mahapatra, 1968; Buckley, 1989; Lovley, 1991). Not all of these ferric oxides are equally suited for reduction by ferric oxide-reducing bacteria. In general, amorphous forms are more efficient for bacterial reduction than the more crystalline forms (Lovley and Phillips, 1986a,b). The combination of high activity of Fe(III) oxides (e.g., amorphous Fe(III) oxides) and low activity of Mn(IV) oxides (e.g., crystalline Mn(IV) oxides) or Fe(III) and Mn(IV) complex oxides with each other often lead to the overlap of Fe(III) and Mn(IV) reduction (Patrick and DeLaune, 1972; Sørensen and Jørgensen, 1987; Patrick and Jugsujinda, 1992) or Fe(III), Mn(IV) simultaneous reduction (Guo et al., 2000).

The reduction of ferric oxide may release phosphate and trace elements that are adsorbed to the ferric oxide (Lovley and Phillips, 1986a). During Fe(III) oxide reduction, some of inorganic phosphorus in the form of ferric phosphate or occluded with insoluble ferric oxyhydroxides can be released when Fe(III) is reduced to more soluble Fe(II) compound (Williams and Patrick, 1973; Barrow, 1983; Sundby et al., 1986). So the reduction of ferric oxide may enhance the availability of these compounds in the soil (Laanbroek, 1990). Fe (III)-reducing organisms can inhibit sulfate reduction and methane production by outcompeting sulfate reducers and methanogens for electron donors (Lovley and Phillips, 1987).

Megonikal et al. (2004) noted that relatively little is known about the abundance of Fe(III)-reducing species *in situ*. Attempts to quantify dissimilatory Fe(III)-reducing bacteria in aquifer sediments suggest that they are dominated by the genera *Geobacter* and *Geothrix* (Anderson et al., 1998; Rooney-Varga et al., 1999; Snoeyenbos-West et al., 2000).

2.6.2 Manganese

Manganese is found in a limited range of mineral deposits, and in variable concentrations in soils, waters and living organisms. The major deposits are sedimentary in origin and consist of carbonates (e.g., rhodochrosite, $MnCO_3$), oxides (e.g., pyrolusite, β-MnO_2; birnessite, δ-MnO_2; hausmannite, Mn_3O_4; and hydroxides (e.g., pyrochroite, $Mn(OH)_2$) (Marshall, 1979). Manganese occurs in a number of valency states, the existence of a particular valency state depending, to a large extent, on the pH and redox potential of the system (Marshall, 1979).

Physico-chemical parameters such as pH and Eh profoundly effect the solubility of Mn in soils (Patrick and Turner, 1968; Mandal and Mitra, 1982; Moore and Patrick, 1989). Mn solubility in aerated soils is extremely pH dependent, with appreciable amounts being brought into solution below pH 5.0 (Adams and Wear, 1957; Gotoh and Patrick, 1972). In the normal pH range (pH 6-9) of natural waters, soluble divalent Mn consists of Mn^{2+} and $MnOH^+$ (Morgan and Stumm, 1965). The solubility of the divalent forms in carbonate-containing waters is governed largely by the solubility of $MnCO_3$ and the pH of the water. Similarly, manganese solubility in sulfide-containing waters is controlled by the solubility product of MnS (Morgan and Stumm, 1965). Mn(III) is thermodynamically unstable and does not occur in soluble form except in the presence of strong complexing agents (Morgan and Stumm, 1965). MnO_2 is the only higher valency form that is thermodynamically stable in natural waters, but the solubility is so low that soluble Mn(IV) is undetectable within the pH range 3-10 (Marshall, 1979).

2.6.2.1 Manganese oxidation

Non-biological oxidation of Mn^{2+} may proceed rapidly at pH values > 9.0 (Van Veen et al., 1978; Ghiorse, 1984). Between pH 7.5 and 9.0, the chemical oxidation of Mn^{2+} is slow, but may be stimulated by hydroxycarboxylic acids. Below pH 7.5, oxidation becomes more and more dependent on microorganisms (Laanbroek, 1990). In acid soils with pH < 5.0, there is hardly any oxidation of manganous ions (Marshall, 1979). Lanbroek (1990) pointed out that it is often very difficult to determine the involvement of bacteria in the oxidation of manganous ions. This is due to the adsorption of Mn^{2+} ions to particulate MnO_2 and the subsequent autooxidation of the adsorbed Mn^{2+} ions (Rosson et al., 1984).

Mn^{2+} oxidizing bacteria can be isolated from various sources, such as soils and aquatic environments (Ghiorse and Hirsch, 1978, 1982). According to Gregory and Staley (1982), 69% of the heterotrophic bacteria are able to oxidize Mn^{2+}. Among these bacteria are species belonging to the genera *Bacillus*, *Caulobacter*, *Chromobacter*, *Pseudomonas* or *Leptothrix* (Laanbroek, 1990; Van Veen et al., 1978). The oxidation of manganese could be summarized as follows (Van Veen et al., 1978; Sikora et al., 2000; Younger et al., 2002; Hallberg and Johnson, 2005):

$$2Mn^{2+} + 2H_2O + O_2 \rightarrow 2MnO_2 + 4H^+ \qquad (2.60)$$

2.6.2.2 Manganese reduction

The reduction of manganese (IV) oxide starts when the redox potential has become sufficiently low (see Table 2-1). Although enrichment cultures of bacteria are able to stimulate the reduction of MnO_2 to Mn^{2+} (Burdige and Nealson, 1985), the direct involvement of bacteria in the reduction of MnO_2 has long been doubtful as MnO_2 may be reduced by sulfide or fatty acids produced by anaerobic bacteria or root exudates such as malic acid (Bromfield, 1958; Jauregui and Reisenauer, 1982, Eq. 2.61). However, it has been observed that direct contact between *Clostridium* sp. and MnO_2 was essential for a rapid reduction (Francis and Dodge, 1988). Also, Lovley et al. (1989) reported that *Alteromonas putrefaciens* was able to grow on hydrogen and MnO_2.

$$2MnO_2 + CH_2O + 4H^+ \rightarrow 2Mn^{2+} + CO_2 + 3\,H_2O \qquad (2.61)$$

Many, but not all, of the microorganisms capable of manganese transformations catalyze similar transformations of iron (Silverman and Ehrlich, 1964). Under anaerobic conditions, Mn^{2+} is quite soluble. Manganese does not readily form an insoluble sulfide phase, which is stable only at high pH, but it may precipitate as rhodocrosite, $MnCO_3$ or it may sorb onto solid-phase $Fe(OH)_3$ or MnO_2 (Morgan and Stumm, 1965; Ponnamperuma et al., 1969; Pasricha and Ponnamperuma, 1976; Wildeman et al., 1993; Hallberg and Johnson, 2005).

2.7 Trace elements

2.7.1 Influence of pH and Eh on trace metal chemistry

Redox potential and pH of the sediment-water system are the major factors known to influence the mobility of trace elements in wetlands. The most important factors are sulfide and Fe/Mn hydrous oxides formation and dissolution (Khalid et al., 1978; Fig. 2-13). Sulfide will under reduced conditions precipitate trace metals resulting in very insoluble compounds while under oxidizing conditions the oxidation of sulfide to sulfate will release these metals into the overlying water (Gardiner, 1974; Engler and Patrick, 1975; see also sections 2.5 and 2.6). On the other hand, hydrous Fe and Mn oxides sorb or co-precipitate trace metals under oxidizing conditions while these metals could be released after Fe/Mn hydrous oxides reduction under reducing conditions (Brooks et al., 1968; Jenne, 1968, references in sections 2.5 and 2.6). If hydrogen sulfide is present, then these metals can form insoluble sulfides. The effect of sulfide and oxides and hydroxides of Fe and Mn in controlling the solubility of trace metals in the sediment-water system is greatly modified by the presence of organics (Morel et al., 1973, Gambrell, 1994). Under anaerobic conditions, organic compounds can bring about reductive dissolution of metal oxides from a higher to a lower oxidation state. This reduction has a dramatic impact on the solubility and speciation of metals. For example, Mn(III, IV), Fe(III), Co(III) and Ni(III) oxides, when reduced to divalent ions under anoxic conditions, show an increase in solubility by several orders of magnitude (Stone and Morgan, 1987). Humic substances, catechols, hydroxyquinones, methoxyphenols, resorcinols, ascorbate, pyruvic acid, oxalic acid, amines, anilines, and other naturally occurring organic compounds, including microbial metabolites, have been shown to have redox reactivity (Stone and Morgan, 1984; Sunda et al., 1983; Waite and Morel, 1984). It is very difficult to single out the specific effects of certain sediment components under such a complex system (Khalid et al., 1978).

Many trace elements such as Cu, Zn, Cd, Pb and Ni are not subject to change in oxidation state as a consequence of soil oxidation-reduction conditions. These are divalent metals under the entire redox potential range encountered in upland and wetland soils (Gambrell, 1994). On the other hand, transformations of some trace elements, such as As, Se or Cr, in the soils and sediments are controlled by oxidation-reduction conditions (Ferguson and Gavis, 1972; Wood, 1974; Hess and Blanchar, 1976; Pierce and Moore, 1982; Neal et al., 1987; Elrashidi et al., 1987; Cooke and Bruland, 1987; Brannon and Patrick, 1987; Oremland et al., 1989; Masscheleyn et al., 1990; 1991a, 1991b, 1992; Masscheleyn and Patrick, 1994; Pardue and Patrick, 1995). Oxidation-reduction processes played a major role in the speciation, solubility, mobility and fate of As, Se and Cr species in wetland soils and sediments. Reduction transformations of all three elements are microbially mediated (Smillie et al., 1981; Masscheleyn et al. 1990, 1991a,b; Fude et al. 1994; Tebo and Obraztsova, 1998; Cervantes et al., 2001). Figure 2-14 shows the critical soil oxidation-reduction potentials for As, Se and Cr redox transformations. Critical redox

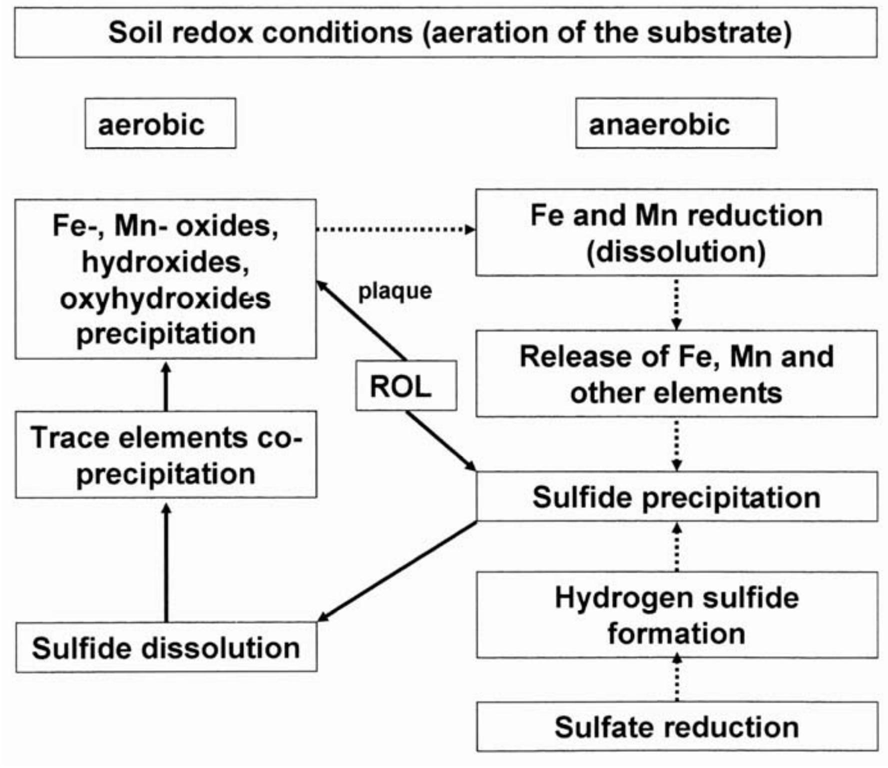


Figure 2-13. Major processes influencing trace elements mobility in wetlands. ROL = radial oxygen loss from plant roots. Solid line = aerobic processes, dashed line = anaerobic processes. Sulfide precipitates may dissolute due to oxygen release from roots.

intensity could be defined as the measured Eh where a change in speciation is observed to occur in a particular soil. The critical Eh is an intensity measurement and will not indicate the extent or rate of the transformation, only that the redox intensity is favorable (Pardue and Patrick, 1995).

At higher soil redox levels (+200 to +500 mV) As(V) is the predominant As species. The reduction of As(V) to As(III) occurs at redox levels corresponding within the nitrate reducing zone of soils characterized by a soil redox level of approximately +200 to +300 mV (DeLaune et al. 1998). Upon reduction, As(III) becomes the major As species in solution, and As solubility increases (Masscheleyn et al., 1991a). Slow kinetics of the As(V) to As(III) reduction coupled with the release of high concentrations of Mn upon reduction made the precipitation of a $Mn_3(AsO_4)_2$ phase under reduced conditions a likely event (Pardue and Patrick, 1995).

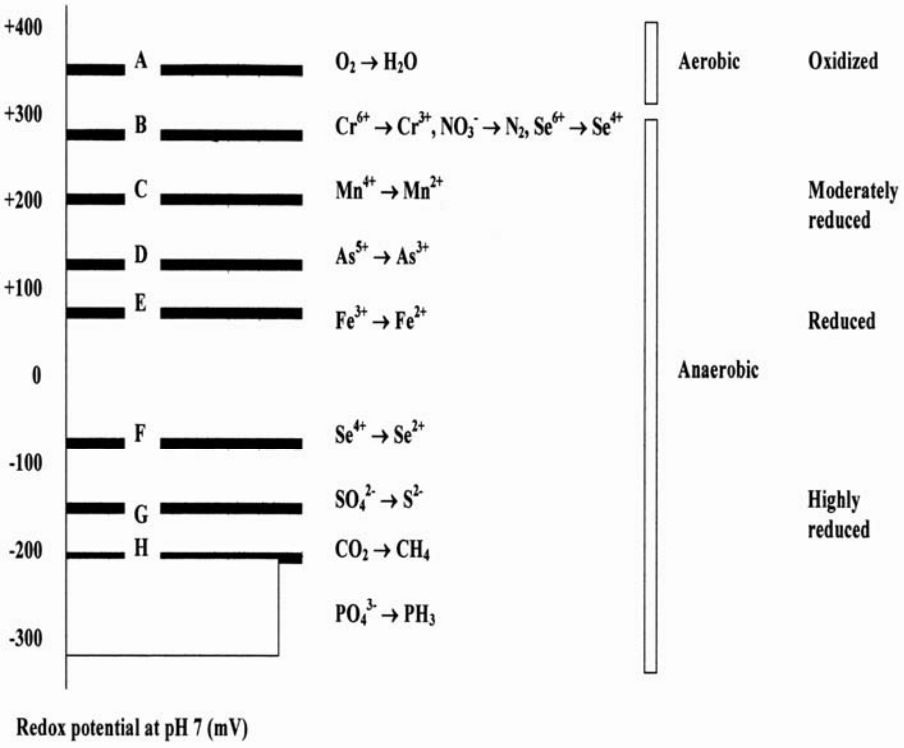


Figure 2-14. Critical redox potential for transformations of various redox couples in wetland sediments. A (Reddy and Patrick, 1975); B (Patrick, 1960; Masscheleyn et al., 1990, 1991b; Buresh and Patrick, 1981); C (Turner and Patrick, 1968); D (Masscheleyn et al., 1991a, b); E (Patrick, 1981; Patrick, 1964); F (Masschelyen et al., 1990), G (Connell and Patrick, 1968), H (Masscheleyn et al., 1992); shaded area, unknown. Modified from DeLaune et al. (1998).

In contrast to arsenic, selenium (VI) is the predominant dissolved species present under high redox conditions (500 mV). Between 200 and 0 mV, Se(IV) becomes the most stable oxidation state of selenium and when Fe reduction starts and soil redox levels drop below +50 mV, selenite (Se^{4+}) is reduced to elemental Se or metal selenides (Se^{2-}) and solubility decreases (Masscheleyn et al., 1991a) due to formation of achavalite (FeSe) and ferroselite ($FeSe_2$) (Mascheleyn et al., 1991c).

Reduction of Cr(VI) occurs at approximately the same redox levels as nitrate reduction. DeLaune et al. (1998) reported that in their experiments, Cr(VI) reduction was complete at 300 mV. The solubility of chromium is strongly dependent upon its oxidation state, Cr(VI) being more soluble than Cr(III). Cr(III), like other cationic metals, is rapidly adsorbed by soil Fe and Mn oxides any clay minerals (Schroeder and Lee, 1975, Bartlett and Kimble, 1976; Dreiss, 1986). Cr(VI), even though not as strongly as Cr(III), is also adsorbed by soil Mn, Al and Fe oxides and clay and mineral colloids (Davies and Leckie, 1980). In contrast to Cr(III), soil redox condition was shown to strongly influence sorption of Cr(VI) from solution (Masscheleyn et al., 1992). Under oxidized soil and moderately reduced (+500 to +100 mV) soil, chromium behavior was dominated by Cr(VI) sorption and reduction to Cr(III). Under more reduced soil redox levels (< +100 mV) chromium chemistry and solubility is controlled by the chemical reduction of Cr(VI) by soluble Fe(II) ions (DeLaune et al., 1998). This abiotically coupled redox reaction is completed within seconds and results in the removal of chromium from solution (Eary and Rai, 1988; Masscheleyn et al., 1992).

2.7.2 Forms of trace elements in wetland soils and sediments

There are a number of general chemical forms of trace elements (including iron, manganese and aluminum) in soils and sediments that differ in their mobility and plant availability. These include (Gambrell, 1994):

1. Water-soluble metals
    a. soluble as free ions, e.g., Zn^{2+}
    b. soluble as inorganic complexes
    c. soluble as organic complexes
2. Exchangeable metals
3. Metals precipitated as inorganic compounds
4. Metals complexed with large molecular-weight humic materials
5. Metals adsorbed or occluded to precipitated hydrous oxides
6. Metals precipitated as insoluble sulfide
7. Metals bound within the crystalline lattice structure of primary minerals

Water-soluble metals are of course the most mobile and plant-available. Exchangeable metals are those bound to soil surfaces by cation exchange process. Metal in this form are considered weakly bound and may be displaced relatively easily to the water-soluble form. Together, metals in the soluble and exchangeable form are considered readily-mobilized and available (Gambrell, 1994). Organic compounds present in the environment can have a significant effect on the solubility and mobility of the metal oxides due to chemical and microbiological action (Francis and Dodge, 1988).

On the other extreme are metals bound within the crystalline lattice structure of clay minerals by isomorphous substitution for some of the common primary cations making up these minerals. Most of these metals are essentially unavailable and would become available only as a consequence of mineral weathering, typically over very long periods of time. Between these two availability extremes are a number of general chemical forms listed above (numbers 3-6) that may be considered potentially mobile and available (Gambrell, 1994).

Precipitation of insoluble compounds and adsorption onto suspended particles can remove metals from the dissolved phase (Davis and Leckie, 1978; Sholkowitz and Copeland, 1981; Stumm and Morgan, 1981). Once associated with the particulate phase, these elements become subject to removal from the water via sedimentation and could be released by various processes of remobilization (Engler et al., 1977; Fejitel et al., 1988). Metals precipitated as inorganic compounds generally include metal oxides, hydroxides, oxyhydroxides and carbonates (Gambrell, 1994; Sheoran and Sheoran, 2006; see also sections 2.6.1.1 and 2.6.2.1). The stability of these inorganic metal compounds is controlled primarily by the system pH, the solubility of the product, and concentrations of the metals and relevant anions (Gambrell, 1994; Sheoran and Sheoran, 2006). At near-neutral to slightly alkaline pH levels, metals tend to be effectively immobilized. If, however, pH becomes moderately to strongly acid, as can sometimes occur when reduced soils or sediments become oxidized, these metals may be released to more mobile forms (Gambrell, 1994).

Iron, aluminum, and manganese can form insoluble compounds through hydrolysis and/or oxidation that occur on wetlands. This leads to formation of variety of oxides, oxyhydroxides and hydroxides Moffet and Zika, 1987; Wieder, 1989; Karathanasis and Thompson, 1995; Tarutis and Unz, 1995; Batty et al., 2002; Woulds and Ngwenya, 2004). Precipitation is greatly influenced by pH. Aluminum can precipitate as Al hydroxides around pH close to 5.0, for Mn and Fe see sections 2.6.1.1 and 2.6.2.1.

Heavy metals may also form carbonates when the bicarbonate concentration in water is high. Although carbonates are less stable than sulfides, they can still perform a significant role in initial trapping of metals (Ramos et al., 1994; Sobolewski, 1996; Sheoran and Sheoran, 2006). Carbonate precipitation is especially effective for the removal of lead and nickel (Lin, 1995):

$$Pb(Ni)SO_4 + Na_2CO_3 \rightarrow Pb(Ni)CO_3 + Na_2SO_4 \qquad (2.62)$$

$$Pb(Ni)Cl_2 + Na_2CO_3 \rightarrow Pb(Ni)CO_3 + 2NaCl \tag{2.63}$$

Co-precipitation is also an adsorptive phenomenon in wetland sediments. The formation of insoluble heavy metal precipitates is one of many factors limiting the bioavailability of heavy metals to many aquatic ecosystems. Iron and manganese oxides and hydrous oxides are important constituents of wetland and estuary sediments. The concentration and distribution of many elements, such as Ni, Cu, Zn or Cd, in sediments and overlying waters are strongly influenced by adsorption and/or co-precipitation with Fe and Mn oxides (Goldberg, 1954; Krauskopf, 1956; Jenne, 1968; Murray and Gill, 1978; Feely et al., 1983; Drever, 1988; Ferris et al., 1989). Copper, nickel, zinc and manganese are co-precipitated in Fe oxides and cobalt, iron, nickel and zinc are co-precipitated in manganese oxides (Stumm and Morgan, 1970). In addition, arsenic and zinc were reported to be retained on iron plaques at the surface of plant roots (Otte et al., 1995). Pardue et al. (1992) found a strong correlation between Al and Pb, Cd and Cr in the coastal Louisiana wetland sediments.

Metal complexes with large molecular weight organics tend to be effectively immobilized. There is some evidence that at least some metals are more tightly bound by organics under anoxic or reducing conditions compared with upland conditions because humic material may become structurally less complex under oxic conditions (Gambrell and Patrick, 1978, 1988, Gambrell et al., 1980, 1991; Guo et al., 1997). However, complex formation with soluble and insoluble organic matter under all conditions of pH and oxidation intensity occurs (Schnitzer and Skinner, 1966; Verloo and Cottenie, 1972).

Most of the heavy metals react with hydrogen sulfide to form highly insoluble metal sulfides (Krauskopf, 1956; Stumm and Morgan, 1981, Kosolapov et al., 2004):

$$M^{2+} + H_2S \rightarrow MS\downarrow + 2H^+ \tag{2.64}$$

where M^{2+} represents a divalent metal ion such as Fe^{2+} (pyrite, FeS_2; pyrrhotite, FeS) , Pb^{2+} (galena, PbS), Cd^{2+} (CdS), Cu^{2+} (covellite, CuS; chalcocite, CuS_2; chalcopyrite, $CuFeS_2$), Ni^{2+} (NiS) or Zn^{2+} (sphalerite, ZnS). These compounds are very stable and insoluble under anaerobic conditions. However, under oxidized conditions sulfides dissolute and release metals (see Eqs. 2.42 or 2.44). This may occur, for example, as a consequence of oxygen release from plant roots in the rhizosphere (Engler and Patrick, 1975; Gambrell et al., 1980; Holmer et al., 1998; Wood and Shelley, 1999; Jacob and Otte, 2003) (see also Fig. 2-13).

Metals bound within the crystalline lattice structure of primary minerals are essentially unavailable in the sedimentary environment. In many literature reports this phase is termed the lithogenic or residual fraction (Gupta and Chen, 1975; Brannon et al., 1976; Khalid, 1980).

2.7.3 Biomethylation

The biomethylation of Hg, Pb, Sn, Te, Cd, As and Se with the production of volatile derivatives such as dimethylmercury, dimethylselenide or trimethylarsine is a well known phenomenon (Craig, 1980; Bentley and Chasteen, 2002). The methylation of metals and metalloids is mediated by a range of aerobic and anaerobic bacteria, as well as fungi, algae, plants and animals, which enzymatically transfer methyl groups to the metals. The methylated compounds formed differ in their solubility, toxicity, and volatility, any may be eliminated from the system by evaporation. Most volatile metal compounds exhibit higher toxicity than their inorganic species since organic derivatives are lipophilic and thus more biologically active (Fatoki, 1997).

Wetlands are favorable to the bacterial methylation of mercury and are known contributors of methylmercury (CH_3Hg) to the environment (StLouis et al., 1994). It has been reported that sulfate reducers are the major biotic contributors to methylmercury formation in marine and freshwater sediments (Trevers, 1986). Mauro et al. (1999) reported that roots of hloating macrophytes such as *Eichhornia crassipes* (Water hyacinth) have been found to be sites of active methylatrion of mercury. Up to 35% of added mercury was methylated and rates were higher at higher temperatures. The methylation was attributed to sulfate-reducing bacteria associated with the roots. Guimaraes et al. (2000) found that roots of the freshwater macrohytes *Eichhornia azurea*, *E. crassipes*, *Paspalum* sp., *Eleocharis sellowiana*, *Salvinia rotundifolia*, *Salvinia* sp. and *Scirpus cubensis* had an order of magnitude higher level of methylation in their roots than the overlying sediments. The methylation was attributed to microorganisms attached to the roots and associated solids, rather than the roots themselves. Biomethylation of Se species is also important under oxidized and moderately reduced conditions (> 0 mV) (Pardue and Patrick, 1995).

2.7.4 Trace elements and wetland plants

Macrophytes have been shown to play important roles in wetland biogeochemistry through their active and passive circulation of elements (Weis and Weis, 2004). Probably two most important soil/sediment factors influencing trace elements mobility in the wetland soils and sediments affected by plants are redox potential (Eh) and pH. Wetland plants influence the redox status in the rhizospehere due to their ability to transport oxygen to the roots (e.g., Armstrong, 1964, 1967, 1971, 1979; Armstrong and Armstrong, 1988, 1990; Armstrong et al., 1990, 1991, 1992; Bedford et al., 1991; Brix, 1990a, 1993b; Sorrell and Armstrong, 1994), which often leads to radial oxygen loss (ROL), that may result in Fe precipitation as iron plaque (see section 2.6.1.1) on one side and dissolution of sulfides (see section 2.5.2) on the other hand. Jacob and Otte (2003) pointed out that the total efflux of ROL varied widely depending on root biomass (Chen and Barko, 1988), density (Pedersen et al., 1995), plant species (Sand-Jensen et al., 1982), location along the root (Connell et al., 1999), root age (Bedford et al., 1991), time of the day (Sand-Jensen et al., 1982) and season. Soil factors such as sulfur, redox-sensitive metals including Fe, Mn, and organic matter content also determine the rate of ROL, since it may depend upon the soil oxygen demand (Sorrell and Armstrong, 1994).

Just as redox potential increases nearer to the soil surface, so does it increase closer to plant roots, and the effect on metal mobility is similar. The precipitation of iron (oxy-)hydroxides in the rhizosphere, due to ROL and the action of microorganisms has been well described and is known to potentially lead to the formation of iron plaque (Jacob and Otte, 2003; see section 2.6.1.1). Not only does this lead to the immobilization of iron, but metals that have high adsorption affinities with such oxides, such as Zn and the metalloid arsenic, may co-precipitate (Otte et al., 1989; St-Cyr and Campbell, 1996). Theoretically, metals and metalloids may become mobilized from associated sulfides upon oxidation in the rhizosphere, but due to their binding affinity may adsorb to iron (oxy-)hydroxidex. This in turn may lead to an influx of metals toward plant roots and accumulation to concentrations higher than in the bulk soil (Jacob and Otte, 2003).

Weis and Weis (2004) concluded that there have been conflicting reports as to whether the presence of the plaque reduces or increases the uptake of metals by the plants. The presence of plaque appeared to reduce the amount of zinc taken up by *Aster tripolium* (Otte et al., 1989) and the amount of manganese taken up by *Phragmites australis* (Batty et al., 2000). The mechanism may have been through the plaque acting as a physical barrier, although the barrier was not effective at low pH conditions. However, in *Typha latifolia*, the presence of iron plaque did not reduce uptake of toxic metals (Ye et al., 1998). Iron plaque increased zinc uptake by *Oryza sativa* and movement into shoots (Zhang et al., 1998). The discrepancies in effects of plaque on metal uptake need to be resolved by further studies. Different metals, environmental conditions or physiologies may account for these differences (Weis and Weis, 2004).

If anoxic/anaerobic conditions remain constant, trace elements including toxic metals, can precipitate out of solution in sulfide-rich sediments, potentially being permanently immobilized (Jacob and Otte, 2003). However, oxygen introduced to such sediments can induce sulfide dissolution, metal release and acid generation (Vigneault et al., 2001). Studies have shown evidence of metal-sulfide oxidation as a consequence of

ROL in the rhizosphere of wetland plants (Howarth and Teal, 1979; Gambrell et al., 1980; Giblin and Howarth, 1984; Holmer et al., 1998).

Rhizosphere pH affects the availability of both soil micronutrients and potentially toxic elements that are not essential for plant growth. With decreasing pH, the availability of essential elements such as zinc, iron, manganese and boron in enhanced by desorption from soil particles. Also potentially toxic aluminum and cadmium availability increases with decreasing pH as a result of increased solubility of these elements (Marschner and Römheld, 1996; Lambers et al., 1998). Molybdenum availability decreases with a decreasing pH, and that of copper, which tends to be complexed with the soil, in unaffected by pH.

The pH of the rhizosphere is greatly affected by the source of nitrogen used by plants because nitrogen is the nutrient required in largest quantities and can be absorbed as either a cation (ammonium) or an anion (nitrate). Roots must remain electrically neutral, so when plants absorb more cations than anions, as when ammonium is the major nitrogen source, more protons must be extruded reducing rhizosphere pH than when nitrate is the major nitrogen source, in which case the pH tends to rise slightly (Lambers et al., 1998).

Metal remobilization may also result from acidification of the rhizosphere by plant exudates (Doyle and Otte, 1997). Roots often excrete exudates that mobilize sparingly soluble micronutrients. Of the plant organic acids, citric- and malic acids have received particular attention (Senden et al., 1995). Synthesis and release of especially citric acid from the roots into the rhizosphere is often considered to increase soil nutrient availability (Brown, 1966; Landsberg, 1981; De Vos et al., 1986; Lipton et al., 1987; Hoffland, 1992). Senden et al. (1995) pointed out that both acids may partly function as a tolerance mechanism under conditions of high element supply. Organic acids are often accumulated in root cell compartments of metal tolerant plant species (Ernst et al., 1975; Mathys, 1977; Thurman and Rankin, 1982; Ojima et al., 1984; Godbold et al., 1984; Kishinami et al., 1987; Miyasaki et al., 1991; Delhaize et al., 1993; Delhaize and Ryan, 1995; Cobbett, 2000).

Generally speaking, accumulation of a given metal is a function of uptake capacity and intracellular binding sites. Figure 2-15. illustrates the processes that are assumed to be influencing metal accumulation rates in plants: mobilization and uptake from the soil, compartmentalization and sequestration within the root, efficiency of xylem loading and transport, distribution between metals sinks in the aerial parts, sequestration and storage in leaf cells. At every level, concentration and affinities of chelating molecules, as well as the presence and selectivity of transport activities, affect the metal accumulation rates (Clemens et al., 2002).

Because of the high binding capacity for metallic micronutrients by soil particles, plants have evolved several strategies for increasing their soil bioavailability. These strategies include the production of metal-chelating compounds such as mugenic and avenic acids (Kinnersley, 1993), which are synthetized in response to iron and possibly zinc deficiencies (Kanazawa et al., 1994; Cakmak et al., 1996; Higuchi et al., 1994, 1996). These chelators were originally called phytosiderophores because of their role in the acquisition of iron (Lambers et al., 1998; Salt et al., 1998). Because it was discovered that these chelators are also important for the uptake of metals like zinc, when these are in short supply, the term phytometallophore would seem more appropriate (Cakmak et al., 1996). In the rhizosphere, phytometallophores chelate and mobilize Fe, Cu, Zn, and Mn (Römheld, 1991). Once chelated to phytometallophores, metal ions can be transported across the plasma membrane as a metal-phytomellophore complex via specialized transporters (Vonwiren et al., 1994, 1995, 1996; Clemens et al., 2002).

The metal absorbed is partitioned between the root system and the shoots. The degree to which roots retain metal, rather than pass it to the shoots, varies greatly with plant species, the metal, and the concentration supplied (Rauser, 1999). Metal transport to the shoot primarily takes place through the xylem (Fig. 2-15). Xylem is a mixed vascular tissue, conducting water and mineral salts taken in by roots throughout the plant, which it provides with mechanical support. In the xylem sap moving from roots to leaves, citrate, nicotianamine, histidine and asparagine are the principal ligands for Fe, Cu, Ni, Co, Mn and Zn (White et al., 1981; Stephan and Scholz, 1993; Senden et al., 1995; Stephan et al., 1996; Kramer et al., 1996; Rauser, 1999). By contrast, chelation with other ligands, such as phytochelatins (enzymatically synthetized cystein-rich peptides) or metallothioneins (cystein-rich polypeptides), might route metals predominantly to root sequestration (Evans et al., 1992).

Metals reach the apoplast (i.e., cell wall continuum) of leaves in the xylem sap, from where they have to be scavenged by leaf cells (Marshner, 1995). Trafficking of metals occurs inside every plant cell, maintaining the concentrations within the specific physiological ranges in each organelle and ensuring delivery of metals to metal-requiring proteins (Clemens et al., 2002). Excess essential metals, as well as non-essential metals are sequestered in leaf cell vacuoles. Different leaf cell types show pronounced differential accumulation with distribution pattern varying with plant species and element (Vögeli-Lange and Wagner, 1990; Clemens et al., 2002, see also Fig. 2-15).

Studies have demonstrated that *Spartina alterniflora* retains metals in standing dead leaves and in detritus (Sanders and Osman, 1985; Kraus et al., 1986). Breteler and Teal (1981) reported increases in Hg, Cu, Fe and Zn concentrations in decomposing *Spartina alterniflora* detritus. The authors considered the increase to be a result of adsorption of metals from the sediment. Zawislanski et al. (2001) reported a rapid the increase in metal

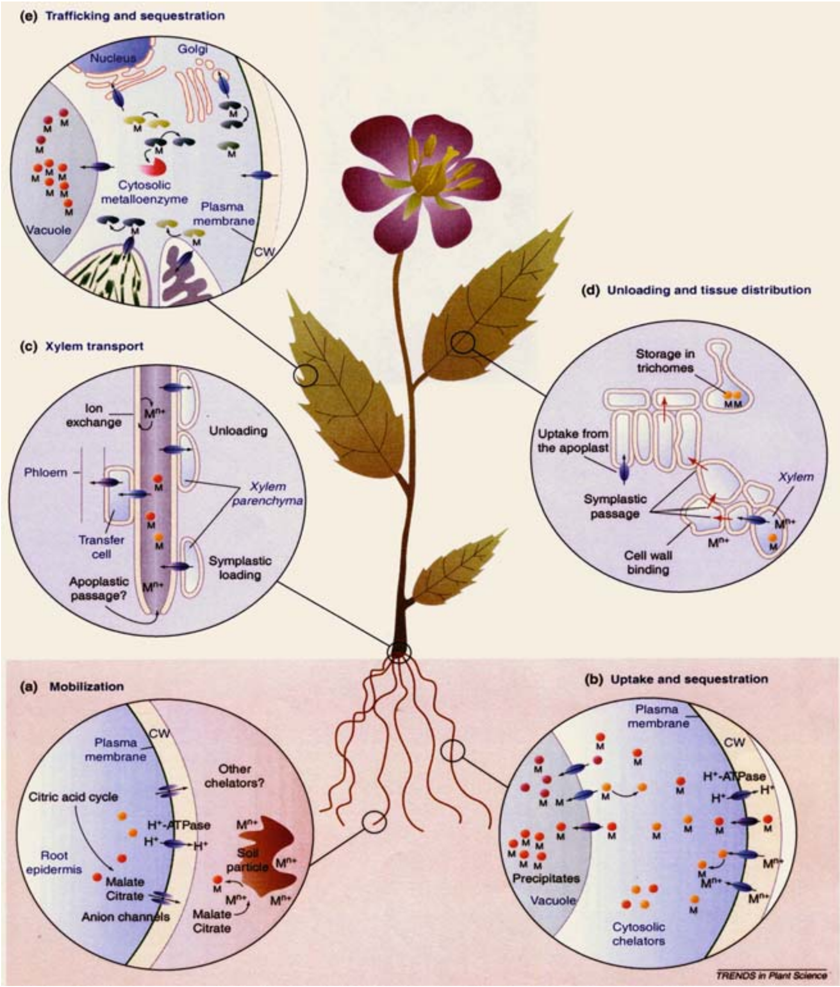


Figure 2-15. Molecular mechanisms proposed to be involved in transition metal accumulation by plants. (a) Metal ions are mobilized by secretion of chelators and by acidification of the rhizosphere. (b) Uptake of hydrated metal ions or metal-chelate complexes is mediated by various uptake systems residing in the plasma membrane. Inside the cell, metals are chelated and excess metal is sequestered by transport into the vacuole. (c) From the roots, transition metals are transported to the shoot via the xylem. Presumably, the large portion reaches the xylem via the root symplast. Apoplastic passage might occur at the root tip. Inside the xylem, metals are present as hydrated ions or as metal-chelate complexes. (d) After reaching the apoplast of the leaf, metals are differentially captured by different leaf cell types and move cell-to-cell through plasmodesmata. Storage appears to occur preferentially in trichomes. (e) Uptake into the leaf cells again is catalysed by various transporters (not depicted in (e)). Intracellular distribution of essential transition metals (= trafficking) is mediated by specific metallochaperones and transport localized in endomembranes in every cell. CW = cell wall, M = metal, filled circles = chelators, filled ovals = transporters, bean-shaped structures = metallochaperones. From Clemens et al. (2002), with permission from Elsevier.

concentrations in decomposing *Spartina foliosa*. The authors hypothetized that the accumulation of fine sediment particles in the litter was responsible for the concentration increase, rather than adsorption from the sediment. Thus despite differences in mechanisms of accumulation, there is a general pattern that decaying litter of marsh plants becomes enriched in metals over time (Weis and Weis, 2004).

Chapter 3

WETLAND PLANTS

3.1 Life forms of wetland plants

Numerous lines of evidence indicate that aquatic angiosperms originated on the land. Adaptation and specialization to the aquatic habitat have been achieved by only a few angiosperms (< 1%) and pteridophytes (< 2%). Consequently, the richness of plant species in aquatic and wetland habitats is relatively low compared with most terrestrial communities (Richardson and Vymazal, 2001). Most are rooted, but a few species float freely in the water (Wetzel, 2001).

Tiner (1999) pointed out that plants growing in wetlands and water are technically called hydrophytes. However, today's usage of the term hydrophyte is different than its original use. In the 1800s and early 1900s, it was used to define aquatic plants that were plants growing in water (Schouw, 1822, as reported in Warming, 1909) or plants with perennating buds beneath the water (Raunikaer, 1905, 1934). Warming and Raunkiaer were among the earliest of the plant ecologists to use the term hydrophyte. Hydrophytes were distinguished from helophytes, which included various wetland plants depending on whose definition was used (Tiner, 1999). Raunkier's life-form were based on a plant's adaptation to the critical season (e.g., winter) mainly the degrees of protection possessed by the dormant buds (Smith, 1913). According to this system, hydrophytes (plants with perennating rhizomes or winter buds) and helophytes (plants with buds at the bottom of the water or in the underlying soil) were the two types of cryptophytes (plants with dormant parts below ground), while other wetland plants were included in other life-forms, such as phanerophytes (trees and shrubs) (Smith, 1913). Raunkiaer's helophytes did not include all typical marsh plants. Tinner (1999) reported that Warming (1909) was probably the first ecologist to arrange plant communities according to the degree of soil wetness. He recognized aquatic plants (water-plants) that spend their entire life submerged or with leaves floating at the surface and terrestrial plants that are mostly exposed to air, including marsh plants. Vegetation was then separated into numerous "oecological classes" based principally on soil properties. The first of the groupings was for soil that was very wet: 1) hydrophytes (formation in water) and 2) helophytes (formation in marsh). Clements (1920) might have been the first ecologist to expand definition of hydrophytes in include helophytes as a type of hydrophyte.

Arber (1920) subdivided the primary groups of rooted and non-rooted aquatic angiosperms according to foliage type, infloresce nse (i.e., flower) type, and whether the leaves and flowering organs are emergent, floating on the water surface, or submersed. The most frequently used simple classification is based on attachment, which has also proven useful in morphological, physiological, and ecological studies (Arber, 1920; Weaver and Clements, 1929; Daubenmire, 1947; Sculthorpe, 1967): 1) emergent macrophytes, 2) floating leaved macrophytes both a) rooted and b) freely-floating, and 3) submerged macrophytes. This classification is used for herbaceous macrophytes, woody plants including both trees and shrubs may form another category.

3.1.1 Emergent macrophytes

Emergent macrophytes are the dominant life form in wetlands and marshes, growing within a water table range from 0.5 m below the soil surface to a water depth of 1.5 m or more (Fig. 3-1). In general, they produce aerial stems and leaves and an extensive root and rhizome system (Brix and Schierup, 1989c). Aerial stems and leaves of emergent macrophytes possess many similarities in both morphology and physiology to related terrestrial plants. The emergent monocotyledons, such as *Phragmites* and *Typha*, produce erect, approximately linear leaves from an extensive anchoring system of rhizomes and roots. The cell walls are heavily thickened with cellulose, which provides the necessary rigidity (Wetzel, 2001).

The root and rhizome systems of these plants exist in permanently anaerobic sediments and must obtain oxygen from the aerial organs for sustained development. Similarly, the young foliage under water must be capable of respiring anaerobically for a brief period until the aerial habitat is reached, since the oxygen content of the water is extremely low in comparison to that of the air. Once the foliage has emerged into the aerial habitat, the intracellular gas channels and lacunae increase in size, thus

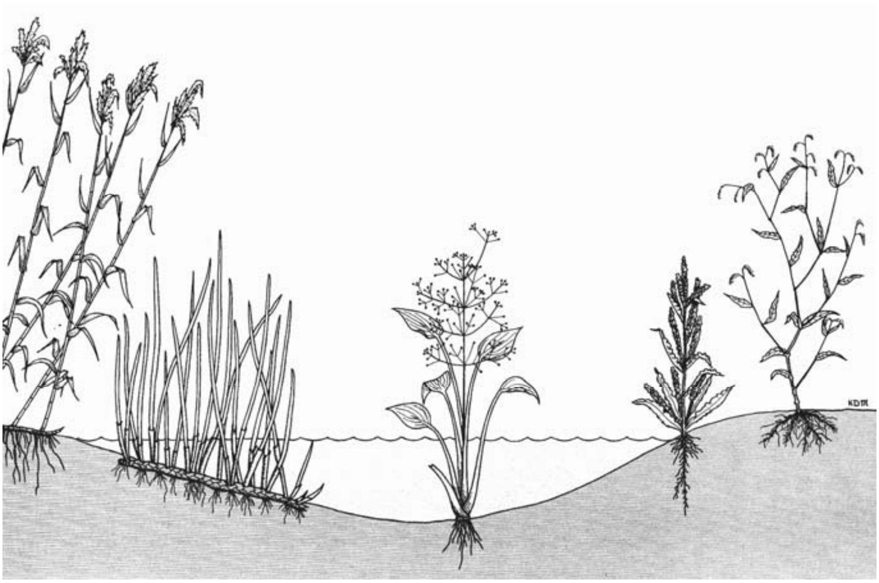


Figure 3-1. Emergent macrophytes (from Sainty and Jacobs, 1981, drawing by David Mackay, with permission).

facilitating gaseous exchange between the rooting tissues and the atmosphere (Wetzel, 2001). These plants are morphologically adapted to growing in a waterlogged or submersed substrate by virtue of large internal air spaces for transportation of oxygen to roots and rhizomes. Part of the oxygen may leak into the surrounding rhizosphere, creating oxidized conditions in an otherwise anoxic environment and stimulating both decomposition of organic matter and growth of nitrifying bacteria (Brix and Schierup, 1989).

Emergent macrophytes assimilate nutrients from sediments and also act as nutrient pumps and play a key role in seasonal changes in available N, P, and K (Gessner, 1955; Wetzel, 1964; Sculthorpe, 1967; Bristow, 1975, Islam et al., 1979; Atwell et al., 1980; Agami and Waisel, 1986). As occurs among terrestrial plants, much (often >90%) of the nutrients used, recycled during growth, and stored in aboveground tissues of wetland macrophytes is translocated back to and stored within rooting tissues belowground (Granéli et al., 1992; DeLucia and Schlesinger, 1995). The release of ions during senescence and decay may be into the water or to the sediment (Denny, 1987) but in both cases the vigorously growing attached microbial community sequesters most of the nutrients being released. The net effect of rooted emergent vegetation is to transfer nutrients from the soil to the surface water via leaching and litterfall, especially at the end of the growing season (Klopatek 1975; Richardson et al., 1978; Richardson and Marshall, 1986). Atmospheric carbon dioxide is clearly a significant source of inorganic carbon among emergent macrophytes (Brix, 1990b; Singer et al., 1994).

Examples of emergent macrophytes include *Acorus calamus* (Sweet flag), *Baumea articulata* (Jointed twigrush), *Bolboschoenus* (*Scirpus*) *fluviatilis* (Marsh clubrush), *Carex* spp. (Sedges), *Cyperus papyrus* (Papyrus), *Eleocharis* spp. (Spikerushes), *Glyceria maxima* (Sweet mannagrass), *Juncus* spp. (Rushes), *Phalaris arundinacea* (Reed canarygrass) *Phragmites australis* (Common reed), *Panicum hemitomon* (Maidencane), *Pontederia cordata* (Pickerelweed), *Sagittaria* spp. (Arrowheads), *Scirpus* spp. (Bulrushes), *Sparganium* spp. (Bur-reeds), *Spartina* spp. (Cordgrasses), *Typha* spp. (Cattails), *Zizania aquatica* (Wild rice)

3.1.2 Submerged macrophytes

Submersed macrophytes occur at all depths within the photic zone, but vascular angiosperms occur only to about 10 m (1 atm hydrostatic pressure). The submerged macrophytes are a heterogenous group of plants (Fig. 3-2). Among the vascular submersed macrophytes, numerous morphological and physiological modifications are found that allow existence in a totally aqueous environment. Stems, petioles, and leaves usually contain little lignin, even in vascular tissues. Conditions of reduced illumination under the water are reflected in numerous characteristics: an extremely thin cuticle, leaves only a few cell layers in thickness, and an increase in the number of chloroplasts in epidermal tissue (Wetzel, 2001).

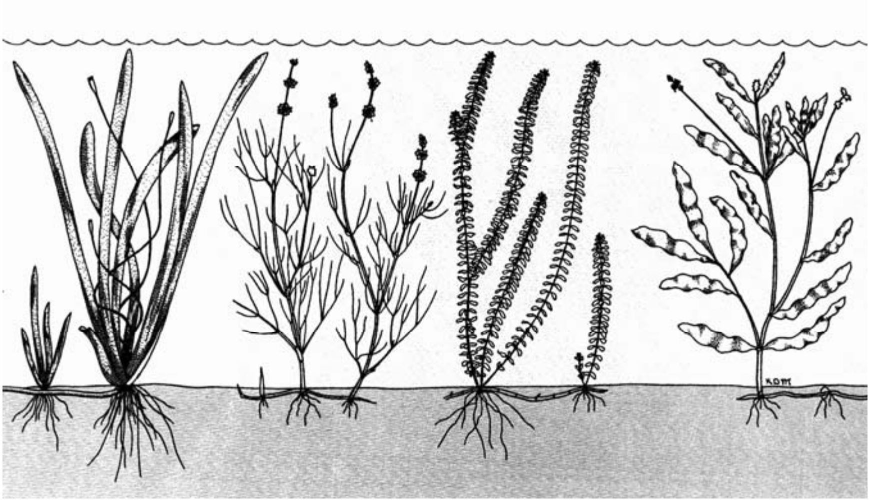


Figure 3-2. Submerged macrophytes (from Sainty and Jacobs, 1981, drawing by David Mackay, with permission).

Rattray et al. (1991) pointed out that interpretation of nutrient uptake and growth in submerged aquatic macrophytes has for a long time been the subject of considerable debate. At present, it is generally accepted that rooted submerged macrophytes can obtain their phosphorus requirements by direct uptake from the sediments (Bristow and Whitcombe, 1971; DeMarte and Hartman, 1974; Best and Mantai, 1978; Bole and Allan, 1978; Carignan and Kalff, 1979, 1980; Welsh and Denny, 1979; Denny, 1980; Barko and Smart, 1980, 1981, 1986; Huebert and Gorham, 1983). However, the absorption of phosphorus from the water is well known as well (Gessner, 1955; Schwoerbel and Tillmanns, 1964a; Bristow, 1975; Denny, 1980; Vermaak et al., 1982; Chambers et al., 1989; Gunnison and Barko, 1989; Rattray et al., 1991). Among most submersed macrophytes, phosphorus absorption rates by foliage are proportional to and dependent on the concentrations in the water. Very large quantities of several mg l^{-1} are rapidly assimilated in excess of requirements until concentrations in the water are reduced to about 10 μg l^{-1}. In most of these studies, however, discrimination between uptake and retention of phosphate by the attached periphyton is not differentiated from that taken into the macrophyte tissue itself (Wetzel, 2001).

Likewise for nitrogen, studies have shown significant uptake from the sediments (Toetz, 1974; Nichols and Keeney, 1976; Best and Mantai, 1978; Barko and Smart, 1981; Barko, 1982; Huebert and Gorham, 1983; Chambers and Kalff, 1987). Rates of nitrate assimilation by foliage of several submersed macrophytes are considerably less than are rates of ammonia assimilation, especially at high pH values (Schwoerbel and Tillmanns, 1964b,c; Toez, 1973, 1974; Cole and Toetz, 1975; Nichols and Keeney, 1976; Holst and Yopp, 1979; Best, 1980; Madsen and Baattrup-Pedersen, 1995). Many submerged macrophytes utilize primarily or only CO_2 as a carbon source, however utilization of HCO_3^- under natural conditions has been reported for a number of submerged species (e.g., Maberly and Spence, 1983, 1989; Wetzel et al., 1985; Ondok and Pokorný, 1987a,b; Bowes and Salvucci, 1989; Madsen and Sand-Jensen, 1991).

Examples of submerged macrophytes include *Cacomba caroliniana* (Fanwort), *Ceratophyllum* spp. (Coontails), *Eggeria densa* (Brazilian waterweed), *Elodea* spp. (Waterweeds), *Hydrilla verticillata* (Hydrilla), *Isoëtes* spp. (Quillworts), *Myriophyllum* spp. (Watermilfoils), *Najas* spp. (Water nymphs, Naiads), *Potamogeton* spp. (Pondweeds), *Utricularia* spp. (Bladderworts).

3.1.3 Floating-leaved macrophytes

3.1.3.1 Free-floating

The freely-floating macrophytes (Fig. 3-3), which occur submersed or on the surface, exhibit great diversity in form and habit, ranging from large plants with rosettes of aerial and/or floating leaves and well-developed submerged roots, e.g., *Eichhornia crassipes* (Water hyacinth) or *Pistia stratiotes* (Water lettuce) to minute surface-floating plants with few or no roots such as Lemnaceae (Duckweeds) *Lemna minor*, *L. gibba*, *L. trisulca*, *Salvinia natans*, *Wolffia* spp. (Brix and Schierup, 1989c). Several of these plants, notably *Lemna*, *Pistia*, *Salvinia* or *Eichhornia* develop so profusely in some waterways and lakes that they inhibit the commercial use of these systems (Hillman, 1961; Moore, 1969). *Eichhornia crassipes* (Water hyacinth) is one of the fastest growing plants in the world.

Freely-floating macrophytes are generally restricted to sheltered habitats and slow-flowing waters. Their nutrient absorption is completely from the water, and most of these macrophytes are found in water bodies rich in dissolved salts (Wetzel, 2001). A few species of freely-floating angiosperms, such as some duckweeds (*Lemna* sp.), utilize both atmospheric and aqueous carbon sources (Wohler, 1966; Wetzel and Manny, 1972; Ultsch and Anthony, 1973; Filbin and Hough, 1985).

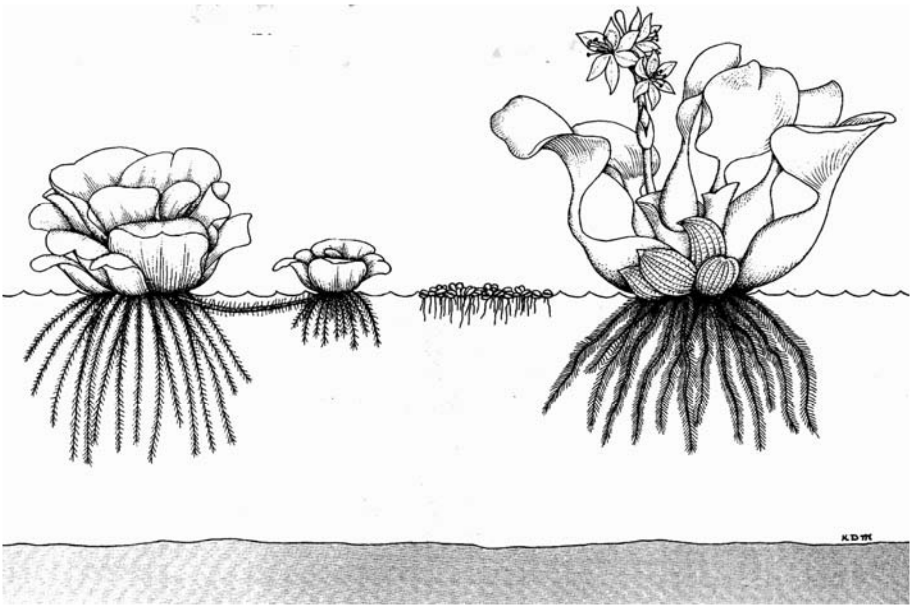


Figure 3-3. Free floating macrophytes (from Sainty and Jacobs, 1981, drawing by David Mackay, with permission).

Most free-floating macropyhtes possess little lignified tissue. Rigidity and buoyancy of the leaves are maintained by turgor of living cells and extensively developed lacunate mesophyll tissue (often > 70% gas by volume). All free-floating rosette plants form well-developed adventitious roots, lateral roots, and epidermal hairs. The root system of the water hyacinth, for example, represents 20 to 50% of the plant biomass (Wetzel, 2001).

3.1.3.2 Rooted

The rooted floating-leaved macrophytes are primarily angiosperms that occur attached to submerged sediments at water depth from about 0.5 to 3.0 m (Fig. 3-4). In heterophyllous species, submersed leaves precede or accompany the floating leaves. Reproductive organs are floating or aerial and floating leaves are on long, flexible petioles (e.g., *Nuphar* or *Nymphaea*), or on short petioles from long ascending stems (e.g., *Brassenia*, *Potamogeton natans*) (Wetzel, 2001). The surface of the water is a habitat subject to severe mechanical stresses from wind and water movements. Adaptations to these stresses by floating-leaved macrophytes include the tendency toward peltate leaves that are strong, leathery, and circular in shape with the entire margin. The leaves usually have hydrophobic surfaces and long, pliable petioles. Despite these adaptations, strong winds and water movements restrict these macrophytes to relatively-sheltered habitats in which there is little water movement (Wetzel, 2001).

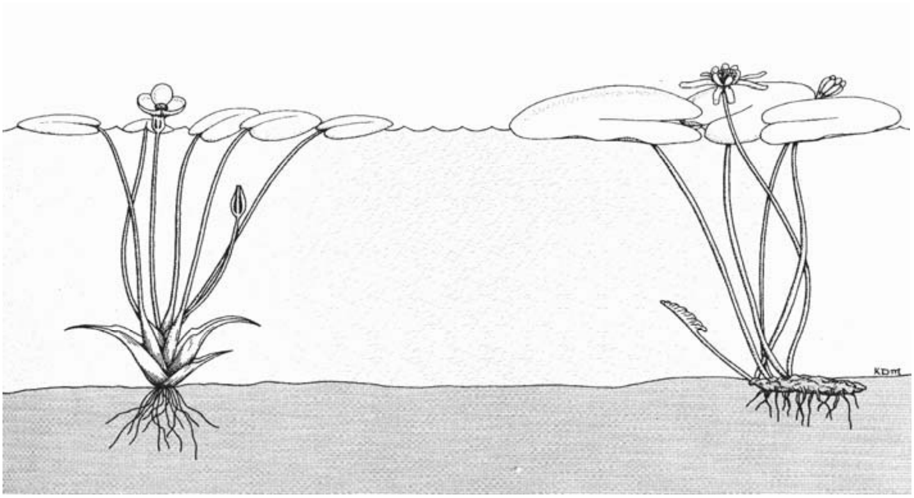


Figure 3-4. Floating-leaved rooted macrophytes (from Sainty and Jacobs, 1981, drawing by David Mackay, with permission).

Floating-leaved rooted macrophytes, particularly the water lilies, exhibit very short leaf longevity averaging about 30 days (50 days during the cooler seasons). Senescence and sinking of leaves are essentially continuous throughout the growing season with an average turnover rate of aboveground foliage of 4-8 times a year (Brock et al., 1983; Kok et al., 1990; Tsuchiya et al., 1990; Kunii and Aramaki, 1992; Kunii, 1999; Carter et al., 2000).

3.2 Plant adaptations to flooding

In well-drained soils, aerobic respiration of plant roots and soil microorganisms under temperate conditions consumes typically between 5 and 24 g oxygen for each square meter of land surface per day during the growing season (Russell, 1973). The upper value, for an agricultural crop at a soil temperature of 17°C, is equivalent to a net daily flux through the soil surface of about 17 liters O_2 m^{-2} at standard temperature and pressure, of which the roots and rhizosphere consume about half (Jackson and Drew, 1984).

With flooding, the soil pore space is totally water-filled, and gas exchange between soil and atmosphere is virtually eliminated (Fig. 3-5.) Because of its low diffusivity in water, the oxygenated zone at the soil surface may be confined to a depth of only a few millimeters (Jackson and Drew, 1984; see also Chapter 2.1).

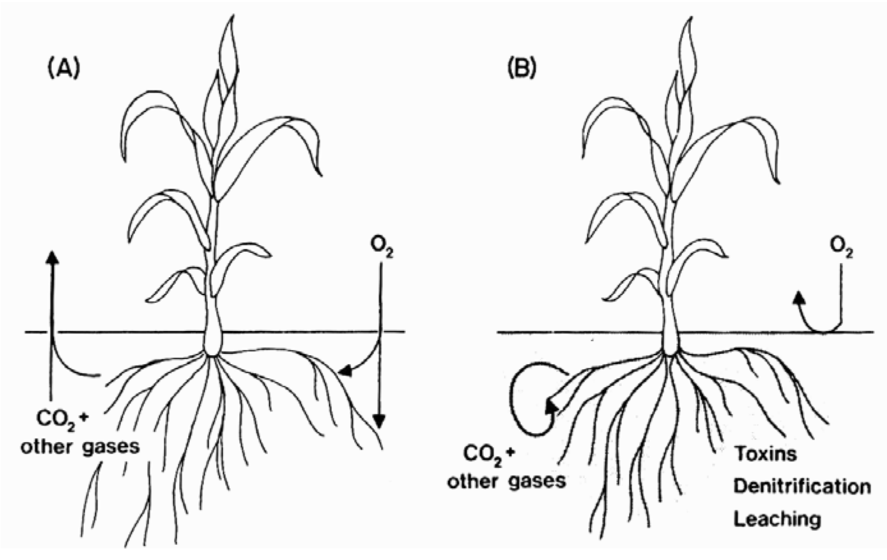


Figure 3-5. Effect of flooding on gas exchange between roots, soil, and atmosphere and on leaching and toxin formation in soil. (A) Well aerated, 3.5 – 17 l O_2 m^{-2} d^{-1}, (B) Flooded. From Jackson and Drew (1984), with permission from Elsevier.

Gopal and Masing (1990) pointed out that flooding may not be viewed as a stress factor of the environment against which the plants must be adapted, but may indeed be an essential requirement for the normal growth. Plants may adapt metabolically to tolerate anoxia (truly anoxia tolerant), adapt morphologically and physiologically to avoid anoxia (apparently anoxia tolerant), or they may not adapt and succumb very quickly to anoxia (anoxia intolerant) (Vartapetian et al., 1978). A few plant species that tolerate prolonged soil flooding exhibit both metabolic and avoidance traits (Hook et al., 1971; John and Greenway, 1976; Vartapetian et al., 1978; Hook and Scholtens, 1978; Davies, 1980; Kozlowski, 1982), which suggests that flood tolerance is not conveyed by a single adaptation but rather by a combination of adaptations (Hook, 1984).

The literature is replete with descriptions of plant responses to flooding (Table 3-1). Such a large number of different responses have contributed to confusion as to which of these are adaptations. However, ability to tolerate or avoid anoxia are fairly common among vascular plants. This suggests there are only a few morphological, physiological, and biochemical solutions to the problem of plant life in a periodically flooded habitat (Hook, 1984).

Another problem in interpreting the plant response to flooding and/or waterlogging is that distinct populations with genotypic or phenotypic differences in flooding tolerance undoubtedly exist (Crawford and Tyler, 1969; Gill, 1970) and in all likelihood, the researchers did not pay attention to this important detail (Tiner, 1999).

A wide range of adaptations make it possible for plants to grow in water or wetlands. These adaptations include physiological responses, morphological adaptations, behavioral responses, reproductive and other strategies. These features and processes also affect the flood and saturation tolerance of species, thereby influencing the distribution of plants within wetlands. The least adapted species posses only minor adaptations and consequently are typically restricted to the margins of the higher elevations. Species with the most effective adaptations are found in the wettest conditions (Tiner, 1999).

The literature concerning responses and adaptations of woody and herbaceous plants to flooding and soil saturation is voluminous and detailed discussion is beyond the scope of this book. Further details on plant adaptations and responses to flooding and waterlogging can be found in Hosner (1960), Gill (1970), Hook and Scholtens (1978), Armstrong (1979), Whitlow and Harris (1979), Crawford (1983, 1987), Hook (1984), Jackson and Drew (1984), Kozlowski (1984a,b,c), Hejný and Hroudová (1987), Hook et al. (1988), Ernst (1990), Gopal and Masing (1990), Jackson et al. (1991), Pezeshki (1994), Brändle et al. (1996), or Tiner (1999).

Table 3-1. Responses of plants to flooding. Based on Hook (1984) with permission from Elsevier).

Response	Selected references
Leaf	
Abscission	Hook (1968), Hook et al. (1971)
Epinasty of leaf and petiole	Kramer (1951), Jackson (1955), Wample and Reid (1975), Kawase (1981)
Petiole reorientation	Kramer (1951)
Stomatal closure	Pereira and Kozlowski (1977), Tang and Kozlowski (1982)
Reduced photosynthesis	Regehr et al. (1975)
Decreased transpiration	Bergman (1920), Parker (1949), Kramer (1951), Regehr et al. (1975)
Increased transpiration	Kramer (1951)
Chlorosis	Kramer (1951), Bergman (1959), Wample and Reid (1975)
Presence of anthocyanin	Parker (1949), Hook (1968)
Thickening of leaf	Hook (1968), Hook et al. (1971)
Decreasing size of leaf	Heinicke (1932), Lindsey et al. (1961), Hejný and Hroudová (1987)
Wilting	Bergman (1920), Kramer and Jackson (1954), Dickson et al. (1965)
Emergence of large leaves above the water surface	Hejný and Hroudová (1987)
Segmentation of leaf laminae	Hejný and Hroudová (1987)
Flower and stem	
Flower abscission	Oskamp and Batjer (1932)
Fruit abscission	Haas (1936)
Corky fruit	Heinicke et al. (1940)
Poor fruit set	Heinicke (1932)
Decreased growth and intermode elongation	Begman (1920), McDermot (1954), Hosner (1960), Hook (1968), Harms (1973)
Increased diameter and height growth	Hook and Brown (1973), Hejný and Hroudová (1987)
Spindly shoot	Heinicke (1932)
Filiformization of the stems	Hejný and Hroudová (1987)
Development of aerenchyma	McPherson (1939), Sifton (1945), deWit (1978), Jackson et al. (1985), Topa and McLeod (1988), Seliskar (1988), Burdick and Mendelssohn (1990), Kludze and DeLaune (1994, 1996)
Transport of oxygen	Van Raalte (1940), Armstron (1979)
Hypertrophy of stem and lenticles	Penfound (1934), Sifton (1945), Hook et al. (1970), Chirkova and Gutman (1972), Drew et al. (1979), Angeles et al. (1986), Topa and McLeod (1988), DeLaune and Pezeshki (1991)
Hollow stems	Billings and Godfrey (1967), Crawford (1983)
Increased moisture stress	Sena Gomes and Kozlowski (1980), Wenkert et al. (1981)

Table 3-1. Cont.

Response	Selected references
Breaking of seed dormancy under conditions favorable for germination	Hejný and Hroudová (1987)
Roots	
Death without replacement	Ye et al. (1969), Hook et al. (1971), Hook and Brown (1973), Wenkert et al. (1981)
Death with replacement with development of new roots	Hook et al. (1971), Coutts and Philipson (1978a,b), Sena Gomes and Kozlowski (1980)
Adventitious roots	Compton (1916), Hosner and Boyce (1962), Gill (1975), Drew and Lynch (1980), Wenkert et al. (1981), Jackson and Drew (1984), Tsukahara and Kozlowski (1985), Van der Sman et al. (1988), Voesenek et al. (1996)
More succulent roots	Kramer (1969), Hook et al. (1971), Keeley (1979)
Shallow roots	Niering (1953), Huenneke (1982)
Fewer root hairs	Snow (1904), Weaver and Himmell (1930)
Increased root and rhizome diameter	Kramer (1969), Cochran (1972), Hejný and Hroudová (1987)
Decreased nutrient uptake	Kramer (1951), Greenwood (1967), Williamson and Splinter (1968)
Increased nutrient uptake	Valoras and Letey (1966), Jones and Etherington (1970), Hook et al. (1983)
Leaking of organic and inorganic compounds from roots	Crawford (1978), Smith and ap Rees (1979), Drew and Lynch (1980), Mendelssohn et al. (1981), Hook et al. (1983)
Accumulation of end products of anaerobic respiration	Crawford (1978), Barclay and Crawford (1982)
Low ethanol production	Crawford (1978)
Increased permeability of plasma membrane and loss of discriminatory uptake of nutrients	Crawford (1978), Smith and ap Rees (1979), Shadan (1980), Hook et al. (1983)
Decreased ectomycorrhizal development	Mikola (1973), Malajczuk and Lamont (1981)
A long-term anabiosis (dormancy) of the underground vegetative organs	Hejný and Hroudová (1987)

3.2.1 Gas transport mechanisms in wetland plants

The major anatomical feature of wetland plants is the presence or development of air spaces in different parts of the leaves, stems, rhizomes and roots (Gopal and Masing, 1990; Brix, 1998; Tiner, 1999). The presence of aerenchyma (air-filled) tissue (Fig. 3.6) and lacunae in many wetlands plants helps these plants to grow in anaerobic or anoxic soils. The extensive lacunal systems which normally contains constrictions at intervals to maintain structural integrity and to restrict water invasion into damaged tissues, may occupy up to 60% of the total tissue volume (Studer and Brändle, 1984). Gopal and Masing (1990) summarized that for long, these air spaces have been considered important reservoirs of oxygen which is transported through well interconnected channels down to the roots and other submerged organs (Arber, 1920; Sifton, 1945, 1957; Williams and Barber, 1961; Kawase and Whitmoyer, 1980).

However, Crawford (1982, 1983) questioned the value of aerenchyma as a reservoir of oxygen for use in roots respiration. He calculated the total air space in several species and the rate of oxygen consumption to show that the total oxygen content in the air spaces would not last for more than a few minutes of the aerobic respiration unless it is renewed rapidly. Due to absence of oxygen in waterlogged soils (see Fig. 3.5 and Chapter 2.1) the roots and rhizomes of plants growing in water-saturated substrates must obtain oxygen from their aerial organs internally through the air spaces in the plants (Laing, 1940; Coult and Vallance, 1951, 1958; Coult, 1964; Teal and Kanwisher, 1966; Armstrong, 1975, 1978, 1979; Dacey, 1980; Dacey and Klug, 1982; Haldemann and Brändle, 1983; Studer and Brändle, 1984).

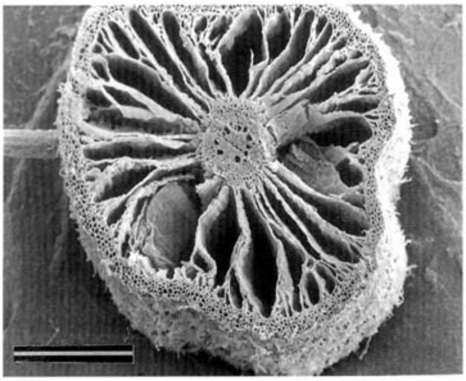


Figure 3-6. Micrograph of gas spaces in roots of *Typha latifolia* (scale bar, 1 mm). From Brix (1998) with kind permission from Backhuys Publishers.

Many studies have demonstrated an increase in aerenchyma in plants subjected to flooding and to stronger anaerobiosis (Seliskar, 1988; Burdick and Mendelssohn, 1990; Kludze et al., 1994; Kludze and DeLaune 1994, 1996). It is not certain whether the formation of aerenchyma is induced by oxygen deficiency or by accumulation of anaerobiosis-caused phytotoxins and gaseous decomposition products (Kludze and DeLaune, 1996). In some species, increased ethylene production during anaerobiosis enhances aerenchyma development (Drew et al., 1979). Aerenchyma formation takes several weeks or months to completely modify root anatomy (Das and Jat, 1977; Keeley and Franz, 1979; Burdick, 1989).

Most of the following text has been adopted from Brix (in Vymazal et al., 1998b, with kind permission of Backhuys Publishers). Internal transportation of oxygen in wetland plants may occur by passive molecular diffusion following the concentration gradients within the lacunal system, and by convective flow (i.e., bulk-flow) of air through the internal gas spaces of the plants (Brix, 1993b). Diffusion is the process by which matter is transported from one part of a system to another as the result of random molecular movement. The net movement of matter will be from sites with lower concentration or partial pressure to sites with lower concentrations (Fig. 3-7). In case of wetland plants, oxygen diffuses along a concentration gradient from the atmosphere into the aerial plant parts and down the internal gas spaces to the rhizomes and roots. Conversely, CO_2 produced by respiration of the root system and CH_4 produced in the sediment diffuse along reverse concentration gradients in the opposite direction (Brix, 1993b).

The internal gas transport mechanism in wetland plants was classically believed to be passive diffusion along the concentration gradients of the individual gases (Evans and Ebert, 1960; Ebert et al., 1962; Teal and Kanwisher, 1966). In *Phragmites australis*, the oxygen concentration is reported to decrease from close to atmospheric levels (20.7%) in the aerial stems to low levels (3.6%) in the lacunal air of the deepest growing rhizomes (Brix, 1988).

The general belief that internal gas transport through the air-spaces of shoots and roots of wetland plants occurred exclusively by gas-phase diffusion has recently been invalidated. In many species convection, i.e., bulk-flow of gas, plays a significant role for aeration of the below-ground tissues. Convective flow of air in plants can be throughflow or non-throughflow (Armstrong et al., 1991) and can be initiated by different physical processes (Brix, 1998). This throughflow mechanism in waterlilies was studied in detail in the early 1980s (Dacey, 1980, 1981; Dacey and Klug, 1982; Grosse and Mevi-Schütz, 1987). The convection in waterlilies is driven by temperature and water vapor pressure differences between the inside of leaves and the surrounding air (Knudsen diffusion) (Schröder et al., 1986; Dacey, 1987). The driving forces are two purely physical processes, namely thermal transpiration and humidity-induced pressurization. Thermal transpiration is the movement of gas through a porous partition where there is a gradient in temperature across the partition. Humidity-induced pressurization is related to pressure differentials induced by differences in water vapor pressure across a porous partition. The result of humidity-induced pressurization is that the total pressure will be higher on the more humid side of the partition. In the case of water lilies, the pressurization is greatest in the youngest leaves.

Pressurized ventilation of the root-system is not restricted to water lilies

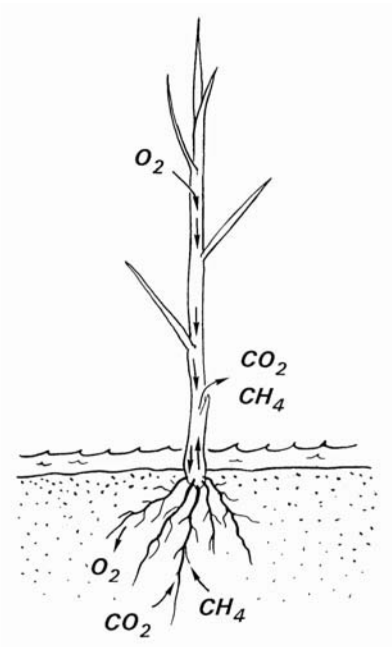


Figure 3-7. Passive diffusion of gases in the lacunal system of wetland plants. Oxygen diffuses along a concentration gradient from the atmosphere into the aerial parts and down the internal gas spaces to the rhizomes and roots. Conversely, CO_2 produced by respiration of the root system and CH_4 produced in the sediment diffuse along the reverse gradient in the opposite direction. From Brix (1993b).

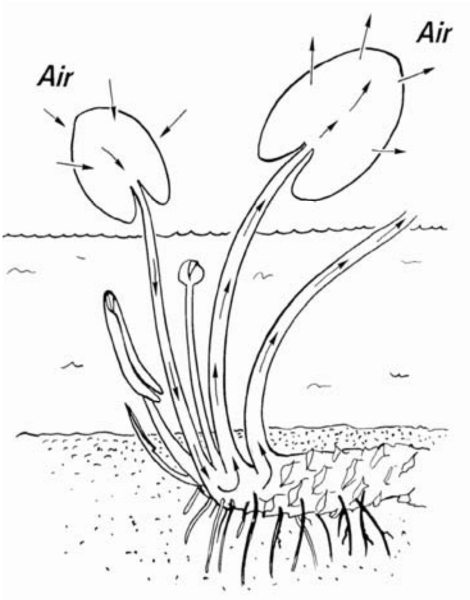


Figure 3-8. Convective gas throughflow in *Nymphaea* spp. Atmospheric air enters the youngest emergent leaves against a small pressure gradient as a consequence of humidity-induced pressurization and thermal transpiration, passes down their petioles to the rhizomes, and up the petioles of older emergent leaves back to the atmosphere. From Brix (1993b).

and species with a similar morphology. The aeration of the rhizomes of *Phragmites australis* (Common reed) is significantly enhanced by a similar convective throughflow mechanism (Armstrong and Armstrong, 1990). Furthermore, investigations have shown that internal pressurization and convective throughflow driven by gradients in temperature and water vapor pressure seems to be common attributes of a wide range of wetland plants, including species with cylindrical and linear leaves such as *Typha* (Cattail), *Schoenoplectus* (Bulrush) and *Eleocharis* (Spikerush) (Brix et al., 1992; Bendix et al., 1994; Tornbjerg et al., 1994).

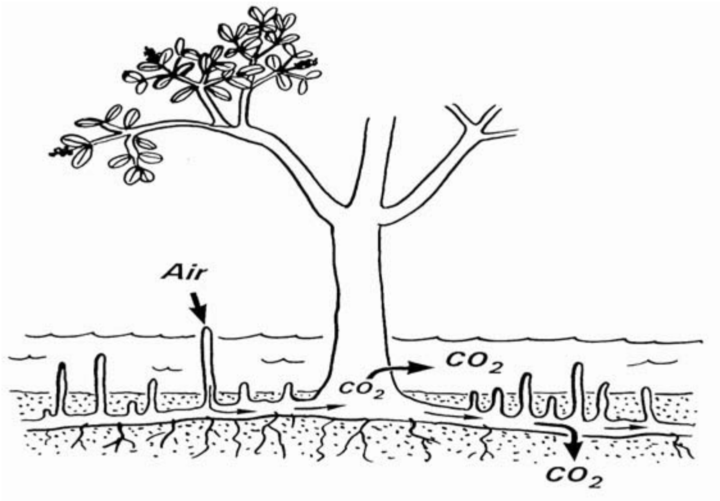


Figure 3-9. Non-throughflow convection in *Avicenia nitida* (Black mangrove) induced by underwater gas exchange. When the tide covers the lenticles on the air roots, the pressure in the air spaces of the roots decreases as a result of respiratory consumption of oxygen and solubilization of respiratory carbon dioxide in the interstitial waters. When the tide falls and the lenticles are again exposed to the air, atmospheric air is drawn into the root system through the first emerging air root. From Brix (1993b). Copyright (©1993) from Constructed Wetlands for Water Quality Improvement by G.A. Moshiri (ed.). Reproduced by permission of Taylor & Francis, a division of Informa plc.

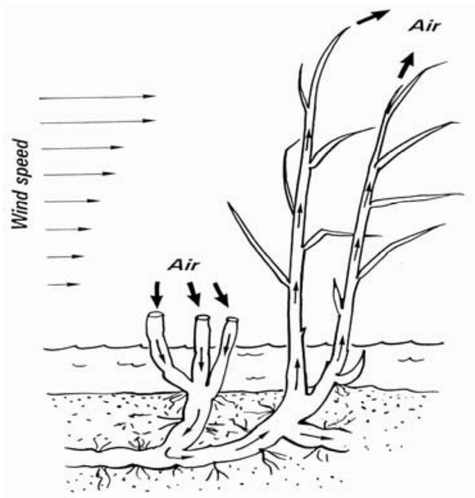


Figure 3-10. Venturi-induced convective throughflow in *Phragmites australis*. The taller shoots and old culms are exposed to higher wind velocities than broken shoots and stubble close to the ground level. This induces a pressure differential which sucks atmospheric air into the underground root systems. From Brix (1993b). Copyright (©1993) from Constructed Wetlands for Water Quality Improvement by G.A. Moshiri (ed.). Reproduced by permission of Taylor & Francis, a division of Informa plc.

Exchange of gases between the gas spaces of buried plant tissues and the surrounding water may also lead to convective air flow inside the plant. This mechanism is based on the different solubility of O_2 and CO_2 in water, CO_2 being about 30 times more soluble in water than O_2. The first observation of convective flow based on underwater gas exchange was observed in *Avicennia nitida* (Black mangrove) in the tidal zone in coastal subtropical areas (Scholander et al., 1955) (Fig. 3-9). The same mechanism of non-throughflow convection has been shown to function in *Oryza sativa* (Deep-water Rice) (Raskin and Kende, 1983, 1985; Stünzi and Kende, 1989).

Another type of convective throughflow – venturi-induced convection (Fig. 3-10) - has been demonstrated in *Phragmites* (Armstrong et al., 1992). This mechanism is based on the gradient in wind velocity around the plant, the velocity being higher at higher positions in the canopy. In contrast to humidity-induced and temperature-induced convection, venturi induced convection can operate in damaged and dead plants and also during the night and winter, where water vapor and temperature gradients are small or lacking (Brix, 1989, 1998).

3.3 Biomass, productivity and decomposition

3.3.1 Biomass

Biomass is most frequently defined as the mass of all living tissue at a given time in a given unit of Earth's surface (Lieth and Whittaker, 1975). It is commonly divided into belowground (roots, rhizomes, tubers, etc.) and aboveground biomass (all vegetative and reproductive parts above the ground level). In plant ecology, the term biomass usually includes all live and dead parts together with litter (Janetschek, 1982). Standing crop includes live parts and dead parts of live plants that are still attached. These dead parts of the plants together with still standing dead plants are called standing litter. Litter refers to those dead parts of the plant that have fallen on the ground or sediment. Peak standing crop is the single largest value of plant material present during a year's growth. In tropical communities, with an almost constant biomass, it is not profitable to search for an annual maximum (Westlake, 1969). However, in all other climatic regions the biomass fluctuates widely throughout the year (e.g., Dykyjová and Květ, 1978; Richardson, 1978; Shew et al., 1981; Vymazal and Kröpfelová, 2005). Květ (1982) gives the following estimates for maximum aboveground standing crop range of different herbaceous plant communities (in g m^{-2} dry mass): submerged (e.g., *Elodea*, *Potamogeton*): 50 – 500; short emergent (e.g., *Carex, Eleocharis*): 300 – 3,000 and tall emergent (e.g., *Phragmites, Typha, Glyceria*): 1800 – 9,900. Wetzel (2001) gives the following values for seasonal maximum standing crop (in g m^{-2}): emergent: 3,980, floating-

Table 3-2. Examples of aboveground standing crop (g m^{-2}) of and net aerial primary productivity (NAPP, g m^{-2} yr^{-1}) of wetland macrophytes growing in natural stands.

Species	Standing crop	NAPP	Location	Ref.
Emergent				
Cyperus papyrus	1,484-7,800	4,180-12,500	Kenya, Uganda	1-5
Glyceria maxima	656-2,690	660-2,860	Germany, Czech Rep., UK	6-13
Phalaris arundinacea	440-2,304	800-2,370	Canada, Czech Rep., India, UK, USA	14-21
Phragmites australis	413-9,890	780-12,898	Australia, Czech R., Denmark, UK, Finland, Germany Morocco, Sweden	22-29
Scirpus lacustris	650-4,200	420-4,600	Czech. R., Denmark, Germany,	8,11, 12, 30
Typha angustifolia	800-5,190	807-4,040	Czech R., Denmark, UK, Germany, Poland,	8, 13, 30 31, 32
Typha latifolia	181-3,560	322-3,560	Canada, Czech R., Poland, UK, USA	13, 19, 33-38
Submerged				
Ceratophyllum demersum	500-700	228-960	Germany, India, Russia, Malawi Sweden,	8, 39, 40, 41, 42
Myriophyllum spicatum	20-402	150-640	Germany, India, USA	8, 43, 44, 45
Potamogeton pectinatus	100-1,313	143-830	Australia, India, Germany, Russia, Netherlands, Poland,	8, 35, 39, 40, 46, 47
Free-floating				
Lemnaceae	10-295	750->1,000	India, Israel	39,48-50
Eichhornia crassipes	440-2,130	200-6,520	India, Japan, USA	39,51-54

1- Thompson et al. (1979), 2- Westlake (1975), 3- Jones and Maturi (1977), 4- Jones (1988), 5- Gaudet (1982), 6- Westlake (1966), 7- Jakrlová (1975), 8- Esteves (1979), 9- Ondok and Dykyjová (1973), 10- Květ and Ondok (1973), 11- Hejný et al. (1981), 12- Véber and Zahradník (1986), 13- Dykyjová (1971), 14- Ho (1979a), 15- Kline and Boersma (1983), 16- Lawrence and Ashford (1969), 17- Hlávková-Kumnacká (1980), 18- Lukavská (1989), 19- Pearsall and Gorham (1956), 20- Handoo and Kaul (1982), 21- Whigham and Simpson (1975), 22- Hocking (1989), 23- Dykyjová and Květ (1982), 24- Larsen and Schierup (1981), 25- Kansanen et al. (1974), 26- Ennabili et al. (1998), 27- Björk (1967), 28- Boar et al. (1989), 29- Rolletschek et al. (2000), 30- Andersen (1976), 31- Mason and Bryant (1975), 32- Kufel (1991), 33- Dykyjová and Květ (1978), 34- Van der Valk and Bliss (1971), 35- McNaughton (1966), 36- Bernatowicz (1969), 37- Jervis (1969), 38- Boyd and Hess (1970), 39- Gopal et al. (1979), 40- Howard-Williams (1979), 41- Dobrokhotova et al. (1982), 42- Forsberg (1960), 43- Kaul et al. (1978), 44- Singh et al. (1982), 45- Adams and McCracken (1974), 46- van Wijk (1988), 47- Royle and King (1991), 48- Porath et al. (1979), 49- Rejmánková (1982), 50- Kaul and Baskaya (1976), 51- Boyd and Scarsbrook (1975), 52- Penfound (1956), 53- Oki et al. (1981), 54- Boyd (1976).

leaved rooted: 850-1,750, free-floating: 150 (Lemnaceae), 1,275 (*Eichhornia*), submersed: 5-140 (nutrient poor), 65-700 (enriched). Examples of standing crop values of wetland plants are shown in Table 3-2.

Roots and rhizomes of submerged macrophytes make up lesser portion of the total plant total biomass than is the case for floating-leaved and emergent species (Table 3-3.).

Table 3-3. Percentage of total biomass found in underground tissues (roots and rhizomes) of mature aquatic macrophytes (based on Wetzel, 2001, with permission from Elsevier).

Type and species	Common name	% of total biomass
Submerged		
Ceratophyllum demersum	Coontail	<10
Elodea canadensis	Common waterweed	<5
Myriophyllum spicatum	Eurasian watermilfoil	6-12
Potamogeton pectinatus	Sago pondweed	18
Potamogeton perfoliatus	Read-head grass	31-51
Vallisneria americana	Wild celery	48
Isoëtes lacustris	Quillwort	20-52
Floating		
*Eichhornia crassipes**	Water hyacinth	10-56
Floating-leaved		
Nuphar spp.	Yellow water lily	46-80
Nymphaea spp.	Water lily	48-80
Emergent		
Acorus calamus	Sweet flag	49-66
Alisma plantago-aquatica	Water plantain	40
Carex lasiocarpa	Hairy fruited sedge	50-78
Carex rostrata	Beaked sedge	18-82
Cyperus fuscus	Brown cyperus	7-8
Cyperus papyrus	Papyrus	31
Eleocharis rostellata	Small-beaked spikerush	47
Equisetum fluviatile	Water horsetail	40-83
Glyceria maxima	Sweet mannagrass	>30-67
Phragmites australis	Common reed	>36-96
Pontederia cordata	Pickerelweed	56-67
Scirpus lacustris	Giant bulrush	>46-90
Sparganium sp.	Bur-reed	>25-66
Typha angustifolia	Narrowleaf cattail	>32-67
Typha latifolia	Broadleaf cattail	29-82
Zizania aquatica	Annual wildrice	7-29

After Westlake (1965, 1966, 1968); McNaughton (1966), Fiala et al. (1968); Knipling et al. (1970); Lack (1973); Bernard (1974); Nicholson and Best (1974); Kansanen and Niemi (1974); Ozimek et al. (1976, 1986); Schiemer and Prosser (1976); Andersen (1976); Szczepańska (1976); Fiala (1976, 1978); Whigham and Simpson (1978); Schierup (1978); Květ and Husák (1978); Kaul et al. (1978); Thompson et al. (1979); Toivonen and Lappalainen (1980); Dykyjová (1980); Grace and Wetzel (1981); Center and Spencer (1981); Han (1985); Twilley et al. (1985); Bernard and Fiala (1986); Hwang et al. (1996); Saarinen (1998). *submerged roots

Biomass is most commonly expressed as dry matter. However, other units are also possible: ash-free dry mass, amount of carbon, amount of assimilated CO_2, amount of released O_2, or amount of energy bound in the mass (Richadson and Vymazal, 2001). The approximate relationships between dry mass and other units were given by Jakrlová (1989):
1 g dry mass (ash < 10%) = 0.9 - 1.0 g organic matter = 0.4 g C = 1.5 g CO_2 = 1.07 O_2 = 17.6 kJ.

Wetzel (2001) gives the following values of ash (as the dry mass percentage): emergent species 12% (5-25%), floating-leaved 16% (10-25%), submerged 21% (9-25)%, average for all species 18%.

3.3.2 Productivity

Gross primary productivity is normally defined as the assimilation of organic matter by a plant community during a specific period, including the amount used by plant respiration. Net primary productivity is the biomass that is incorporated into a plant community during a specified time interval, less that respired. This is the quantity that is measured by harvest methods and which has also been called *net assimilation* or *apparent photosynthesis*. *Net aerial* (or *aboveground*) *primary production* (NAPP) is the biomass incorporated into the aerial parts (leaf, stem, seed, and associated organs) of the plant community (Milner and Hughes, 1968). NAPP is an important variable for the analysis of energy and nutrient flows in wetlands. Estimation of NAPP has been a difficult task because it can only be assessed indirectly from the storage and flows of biomass in an ecosystem (Hsieh, 1996). The detailed description of NAPP measurement is beyond the scope of this book and reader may find necessary information in Wiegert and Evans (1964), Milner and Hughes (1968), Bradbury and Hofstra (1976), Shew et al., (1981), Dickerman et al. (1986), Kaswadji et al. (1990), de Leeuw et al. (1996) or Richardson and Vymazal (2001).

Due to strong seasonality of climate and because some of the water bodies are temporary, a large number of macrophytes behave like annuals, i.e., they exhibit a unimodal growth with a single peak of biomass. Thus, the peak standing crop (or the difference between the peak and lowest standing crop if the plants do not die completely) during the growth period can be theoretically considered equivalent to the net production (Vyas et al., 1990). However, this often represents an underestimation because substantial amounts of plant parts are lost in death and predation. In submerged and free-floating species with rapid vegetative multiplication, death of individual parts or plant parts accounts for very large losses. The estimation of belowground production in emergents and other rhizomous plants remains an unsolved problem. In fact, most productivity data are estimates computed by different methods and are not the real measurements (Vymazal, 1995a). Examples of NAPP of wetland plants are given in Table 3-2.

Westlake (1963) estimated following average productivity (g m^{-2} yr^{-1}) of various plant communities: submerged macrophytes: 600 (temperate climate) and 1,700 (tropical); salt marsh: 3,000; reedswamp (i.e. *Phragmites* marsh) 4,500 (temperate) and 7,500 (tropical). Gopal and Masing (1990) listed following values for primary productivity of various plant groups (in g m^{-2} yr^{-1}): tall emergent species, 500 to 2,000; and short emergent species, 200 to 1500. Květ (1982) estimated the net primary productivity as follows (g m^{-2} yr^{-1}): submerged species (e.g. *Elodea, Potamogeton*): 100 - 500; short emergent species (e.g. *Eleocharis, Bolboschoenus*): 500 – 3,000; tall emergent species (e.g. *Phragmites, Typha, Glyceria*): 1,400 – 4,800; tall sedges and grasses (graminoids, e.g. *Carex*, *Calamagrostis*): 1,000 – 3,100.

Gopal and Masing (1990) pointed out that productivity of submerged species is often much greater than the maximum standing crops because of mortality and herbivory. Using different methods production of submerged species has been estimated to be from 15% to 60% greater than the maximum biomass (Ikusima, 1966; Dawson, 1976; Sand Jensen and Sondergaard, 1978, Howard-Williams, 1978; Carpenter, 1980). A similar situation occurs for free-floating macrophytes. For example, standing crop values for Water hyacinth (*Eichhornia crassipes*) have aften been reported between 2,000 and 6,000 g m^{-2} while annual production could be expected between 4,500 and 9,000 g m^{-2} g m^{-2} yr^{-1} (Yount and Crossman, 1970; Boyd, 1976; Westlake, 1963; Wooten and Dodd, 1976; Center and Spencer, 1981; DeBusk et al., 1981). Květ (1982) gives the estimation of turnover coefficients (i.e., net primary productivity / maximum season standing crop) for these communities: submerged species: 1.1 - 1.5, short emergent species: 1.05 - 1.5, tall emergent species: 1.05 - 1.3, tall sedges and grasses (graminoids): 1.15 and phytoplankton: 450 - 600.

3.3.3 Decomposition

Decomposition generally refers to the disintegration of dead organisms into particulate form and the further breakdown of large particles to smaller and smaller particles, until the structure can no longer be recognized and complex organic molecules have been broken down into CO_2 and H_2O and mineral components (Mason, 1977). In wetland studies, the term decomposition is mostly confined to the breakdown and subsequent decay of dominant macrophytes which leads to the production of detritus (Vymazal, 1995a).

Most of the annual aboveground production in wetlands is not consumed by herbivores but decomposes on the wetland surface (Gallagher, 1978; Polunin, 1982). Rates of decomposition in wetlands vary greatly and the fate of materials released and adsorbed during decomposition depends on the physical and chemical composition of materials as well as environmental conditions at the site of decomposition (Willoughby, 1974; Saunders, 1976;

Saunders et al., 1980; Brinson et al., 1981; Godshalk and Barko, 1985; Gopal and Massing, 1990; Vymazal, 1995a).

Decomposition involves several steps forming a part of the long continuum in time starting with senescence and passing through the stages of death and gradual disintegration to the release of various elements in different forms (Gopal and Massing, 1990). Three fundamental processes are involved, occurring simultaneously. Firstly, soluble substances (e.g., Na^+, K^+, Ca^{2+}, Mg^{2+}, N, P, labile organic compounds such as sugars, fatty acids or amino acids) are leached out in water by abiotic leaching, either with runoff from senescent and standing dead organs, or after dead shoots or leaves have fallen and come into contact with water. Such release may begin by autolysis even before the death of the tissue (Golterman, 1973; Otsuki and Wetzel, 1974). This process is quite rapid and accounts for majority of mass reduction during the early stages of decomposition with most of the water-soluble organic substances being released within 6 to 12 months. The rapid initial release of nutrients by leaching has been documented in many marsh plants – up to 30% of nutrients are lost by leaching alone during the first few hours or days of decomposition (Cummins et al., 1972; Wetzel and Manny, 1972a; Petersen and Cummins, 1974; Davis and Van der Valk, 1978; Boyd, 1970b; Mason and Bryant, 1975, Polunin, 1982, Esteves and Barbieri, 1983; Howard-Williams et al., 1983). In submerged and floating-leaved plants, leaching accounts for up to 50% loss of dry matter within first 2-3 days (Gopal, 1990).

The second process involved in plant biomass decomposition is mechanical fragmentation of dead parts into smaller bits of various size by the action of wind, wave and tidal action, or animal trampling, biting or cutting (de la Cruz, 1979). This is accompanied by the third process, a microbial oxidation of organic matter mediated by the activity of bacteria and fungi.

Wetland decomposition may be either aerobic or anaerobic. Aerobic decomposition occur (de la Cruz, 1979: 1) in the air, while dead plants are still standing or while dead plant tissues are still attached to living plants; 2) in the water, after dead plant parts are carried by wind or precipitation or simply dropped into the water flowing over a wetland (depending on their size and moisture saturation, these plant parts may either float on the surface, become suspended in the water column, or sink to the bottom; and 3) on the wetland floor. The depth and extent of the anaerobic zone depend on the type of rooted aquatic vegetation, the nature of the rhizosphere, the activity of the benthic organisms and the other physical conditions of the substrate (de la Cruz, 1979). Anaerobic decomposition is much slower than decomposition under aerobic conditions (Williams and Gray, 1974; Godshalk and Wetzel, 1978a) and is done by obligate anaerobic and some facultative anaerobic bacteria (Chamie and Richardson, 1978). In addition, anaerobic decomposition generally does not proceed to completion owing to the lack of oxidizing power and adequate levels of alternate electron acceptors to carry on the process.

A common assumption, made earlier in terrestrial studies, and widely used in wetland studies, is that decay rate is a constant proportion of the amount remaining, i.e., it decreases logarhitmically (Jenny et al., 1949, Olson, 1963). Jenny et al. (1949) formulated non-constant integrative parameters, known as k and k′, to denote the decomposition rate of vegetation under steady-state conditions. Simply stated, k′ equals the percentage loss of the original weight over a specified period of time with a single year usually used. The term k equals the –ln(1-k′) or the negative natural logarithm of the percentage weight remaining after a designated period of time (Chamie and Richardson, 1978):

$$\ln (W_t/W_o) = -kt \quad \text{or } W_t = {}'W_o e^{-kt} \tag{3.1}$$

where W_o is the dry mass initially present, W_t is the dry mass remaining at the end of the period of measurement, and t is time in years. Using this expression, the time for 50% decomposition is calculated as 0.693 k^{-1} and for 95% decomposition as 3 k^{-1}. This equation does not account for rapid leaching losses, and therefore, large variations are introduced into the calculated rates based on different study periods but similar leaching losses (Gopal, 1990). However, these decay coefficients are used routinely to compare rates of decomposition (Godshalk and Barko, 1985). More sophisticated models usually in some way take into account the fact that detritus is not homogenous, and that rates of decay, i.e., percent loss per fay, decrease through time (Godshalk and Wetzel, 1978b; Carpenter 1981, 1982).

Godshalk and Barko (1985) estimated the average decay constant k (d^{-1}) for submersed-floating and emergent marsh vegetation to be 0.008 and 0.0031, respectively. For emergent vegetation, it results in 50% and 95% weight loss in about 220 days and 970 days, respectively. For comparison, deciduous leaves, conifer leaves and wood the average k (d^{-1}) was 0.0064, 0.0049 and 0.00013, respectively.

3.4 Evapotranspiration

Atmospheric water losses from a wetland occur from the water and soil – evaporation, and from emergent portions of plants – *transpiration*. The combination is termed evapotranspiration. It has been known for a long time that evapotranspiration is a very important component of the water balance and ecology of wetland ecosystems (e.g., Otis, 1914; Penfound and Earle, 1948; Timmer and Weldon, 1967). Evapotranspiration to open water evaporation ratio (E/E_o) is commonly >1.0 and may reach values up to 5 or more (Table 3-4).

Table 3-4. Representative rates of evapotranspiration (E_t) by aquatic plants and comparison to rates of evaporation from open lake water (E_o). Based on Wetzel (2001) with permission from Elsevier. Emergent species unless marked: FF – free floating. FL – floating leaved.

Species (locality)	**mm d^{-1}**	**E_t/E_o**	**Reference**
Alternanthera philoxeroides (Alabama)	4.0 – 6.3	1.26	1
Arctic bog (Canada)	2.2 – 7.3		2
Azolla caroliniana (Argentina) FF	7.1	0.95	3
Carex acutiformis/Sphagnum (The Netherlands)	1.0 – 3.7	1.65	4
Carex diandra (The Netherlands)	1.1 – 3.9	1.68	4
Carex lurida (Alabama)	4.0 – 6.3	1.33	1
Carex dominated subarctic marsh (Ontario)	2.6 – 3.1	0.74 – 1.02	5
Eichhornia crassipes (Alabama) FF	6 - 11	1.31 – 2.52	6
E. crassipes (India) FF	3.8 – 10.5	1.30 – 1.96	7
E. crassipes (Argentina) FF		2.67	3
E. crassipes (Florida) FF		1.50 – 2.52	8
E. crassipes (Texas) FF		3.20 – 5.30	9
Juncus effusus (Alabama)	3.8 – 8.0	1.52	1
Justicia americana (Alabama)	2.2 – 6.4	1.17	1
Marsh grasses (Czech Republic)	2.0 – 10.5		10
Nymphaea lotus (India) F-L	2.5 – 6.0	0.82 – 1.35	7
Oryza sativa (Australia)	6 - 13		11
Panicum regidulum (Alabama)	5.5 – 7.5	1.58	1
Phragmites australis marsh (Czech Republic)	1.4 – 6.9	1.03	12
P. australis marsh (Czech Republic)	6.9 – 11.4		13, 14
Salvinia molesta (India) FF	2.1 – 6.8	0.96 – 1.39	7
Sedge-grass marsh, *Carex, Calamagrostis, Glyceria* (Czech Republic)	2.2 – 4.5		15, 16
Typha domingensis (Florida)	2.7 – 4.7	1.3	17, 18
Typha latifolia (Alabama)	4 – 12	1.05 – 2.50	6
T. latifolia (The Netherlands)	0.9 – 4.7	1.87	4
T. latifolia (Poland)		1.20 – 2.40	19
Typha sp. marsh (Czech Republic)	3.2 – 5.7		20
Willow carr, *Salix* sp. (Czech Republic)	2.4 – 4.8		16

1- Boyd (1987), 2-Roulet and Woo (1988), 3-Lallana et al. (1987), 4-Koerselman and Beltman (1988), 5-Lafleur (1990), 6- Snyder and Boyd (1987), 7-Rao (1988), 8-De Busk et al. (1983), 9-Idso (1979), 10-Rychnovská et al. (1972), 11-Humphreys et al. (1994), 12- Šmíd (1975), 13-Květ (1973), 14-Rychnovská and Šmíd (1973), 15-Přibáň and Ondok (1985), 16-Přibáň and Ondok (1986), 17-Glenn et al. (1995), 18-Abrew (1996), 19- Bernatowicz et al. (1976), 20-Přibáň and Šmíd (1982).

Evaporative losses from water bodies and wetlands are greatly modified by the transpiration from emergent and floating-leaved aquatic plants. Rate of transpiration and evaporative losses to the atmosphere vary with an array of physical (e.g., wind velocity, humidity, and temperature) and metabolic parameters and structural characteristics of different plant species (Brezny et al., 1973; Bernatowicz et al., 1976, Přibáň and Ondok, 1985, 1986; Boyd, 1987; Koerselman and Beltman, 1988; Jones, 1992).

Evapotranspiration play also an important role in constructed wetlands for wastewater treatment. Some treatment systems use plant species with high rate of evapotranspiration, namely willows (*Salix* sp.) in order to provide zero discharge (for details see section 4.4).

3.5 Role of macrophytes in constructed wetlands

The presence of macrophytes is one of the most conspicuous features of wetlands and their presence distinguish constructed wetlands from unplanted soil filters or lagoons. The macrophytes growing in constructed wetlands have several properties in relation to the treatment process (Table 3-5) that make them an essential component of the design (Brix, 1996).

Table 3-5. Summary of the role of macrophytes in constructed treatment wetlands. From Brix (1996) with permission from Universität für Bodenkultur, Vienna, Austria.

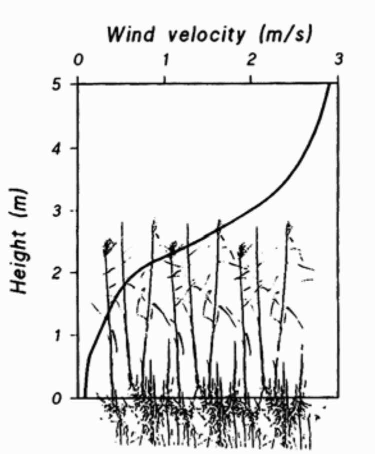
Macrophyte property	**Role in treatment process**
Aerial plant tissue	Light attenuation → reduced growth of phytoplankton Influence of microclimate → insulation during winter Reduced wind velocity → reduced risk of resuspension Aesthetic pleasing appearance of the system Storage of nutrients
Plant tissue in water	Filtering effect → filter out large debris Reduced current velocity → increased rate of sedimentation, reduced risk of resuspension Provides surface area for attached biofilms Excretion of photosynthetic oxygen → increases aerobic degradation Uptake of nutrients
Roots and rhizomes in the sediment	Stabilizing the sediment surface → less erosion Prevent the medium from clogging in vertical flow systems Release of oxygen increase degradation (and nitrification) Uptake of nutrients Release of antibiotics

The presence of vegetation in constructed wetlands distributes and reduces the current velocities of the water (Pettecrew and Kalff, 1992; Somes et al., 1996). This creates better conditions for sedimentation of suspended solids and reduces the risk of resuspension. Stands of emergent macrophytes substantially reduce the wind velocities near the soil or water surface (Fig. 3-11) as compared to velocities above the vegetation (Brix, 1994a, 1998). This creates suitable conditions for settlement of suspended solids, prevents resuspension, thus improving the removal of suspended solids in constructed wetlands with free water surface. On the other hand, lower wind speed reduces aeration of the water column.

Macrophytes attenuate the light penetration into the water column thus limiting the algal growth. Macrophyte shading on phytoplankton is expected to be particularly important in shallow habitats where macrophyte cover is extensive. In the case of free-floating macrophytes, such as Water hyacinth or Duckweed, which can cover completely the surface of the wetlands, algal growth is limited to minimum due to lack of light. This is desirable as in constructed wetlands, phytoplankton growth is not appreciated because of increase of suspended solids in the outflow.

Another important effect of the plants is the insulation that the cover provides during winter (Fig. 3-11) especially in temperate and cold climatic regions (Smith et al., 1996; Mander and Jenssen, 2003). When the standing litter is covered by snow it provides a perfect insulation and helps keep the soil free of frost (Brix, 1998). The litter layer also helps in protecting the soil from freezing during winter, but on the other hand, it also keeps the soil cooler during spring (Haslam, 1971a, b, Brix, 1994a). The insulation is particularly important in constructed wetlands with sub-surface flow.

In vertical flow constructed wetlands, where wastewater is fed onto the bed surface, the presence of macrophytes helps to prevent clogging of the medium (Bahlo and Wachs, 1990). The movement of plants as a consequence of wind, etc., keep the surface open for water percolation by creating annular holes in the surface around the stems.



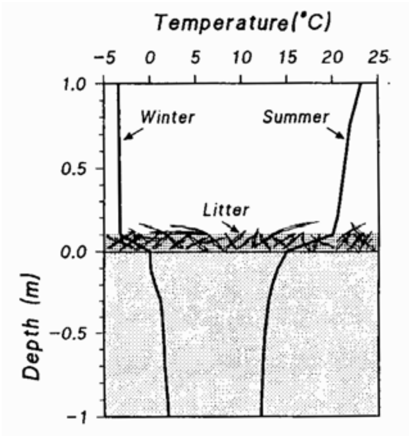


Figure 3-11. Effect of a dense canopy of *Phragmites australis* on the wind velocity (left) and effect of litter layer within a stand of *P. australis* on the soil temperature during winter and summer, respectively. From Brix (1998) with permission from Backhuys Publishers.

The stems and leaves of macrophytes that are submerged in the water column provide a huge surface area for biofilms (Gumbricht, 1993 a,b; Chappell and Goulder, 1994). The submerged plant tissues are colonized (Fig. 3-12) by dense communities of photosynthetic algae as well as bacteria and protozoa (Brix, 1998). On one side the algae oxygenate the water and take up nutrients, bacteria degrade organics but on the other hand, thick periphyton may limit the growth of submerged macrophytes through absorption of PAR (photosynthetically active radiation) before reaching the leaf surface of vascular plants. This may cause even the macrophyte decline (Phillips et al., 1978; Sand-Jensen and Søndergaard, 1981; Sand-Jensen and Borum, 1983, 1991; Twilley et al., 1985).

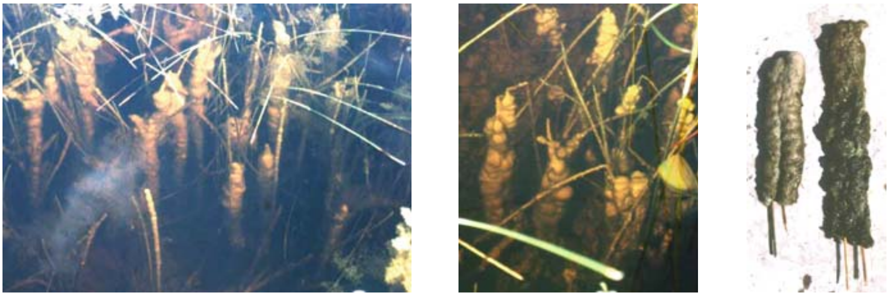


Figure 3-12. Periphyton growing on stems of *Eleocharis cellulosa* (Spikerush) and *Utricullaria* spp. (Bladewort) in the Florida Everglades (left and middle). Right-detail of the periphyton sheaths growing on *Eleocharis*. Photos by Jan Vymazal

The belowground biomass plays an important role in sub-surface constructed wetlands. The claim that hydraulic conductivity of any soil in sub-surface flow CWs will improve after the period of three years and once developed, it will stabilize and maintain itself due to channels created by the living and dead roots and rhizomes (Kickuth, 1981) failed in real systems (Haberl and Perfler, 1990; Schierup et al., 1990a,b). On the contrary, the hydraulic conductivity quite often decreases over the period of operation (McIntyre and Riha, 1991; Marsteiner et al., 1996).

Roots and rhizomes of macrophytes provide a substrate for attached microorganisms (Hoffmann, 1986). Also, Paul and Clark (1996) pointed out that total microbial counts increase significantly in the rhizosphere. Total number of bacteria in the distance from the root of up to 1 mm is usually 10 times higher as compared to the zone located 15-20 mm from the root surface. Both calculations and direct microscopy show that bacterial coverage of the root surface usually ranges from 5 to 10%. Single bacteria are often associated with pits in the root cell walls and clusters of bacteria at cell junctions. The plants are known to be able to affect rhizosphere populations (Paul and Clark, 1996).

Wetland plants require nutrients for growth and reproduction. As wetland plants are highly productive (see Table 3-2), considerable amounts of nutrients can be bound in the biomass (for emergent plants see section 2.3.8 for nitrogen and section 2.4.2.4 for phosphorus). The highly productive

Eichhornia crassipes (Water hyacinth) has higher uptake capacity and nutrient standing stocks may reach up to 230 g N m^{-2} and 30 g P m^{-2} whereas nutrient standing stocks of submerged macrophytes is much lower (<20 g N m^{-2} and < 3 g P m^{-2}) (Vymazal, 1995a; Brix in Vymazal et al., 1998b). However, the amounts of nutrients that can be removed by harvesting is generally insignificant compared to the loading into constructed wetlands for wastewater treatment (Vymazal, 2004, 2005a, see also sections 5.4.3 and 5.4.4).

It is well documented that aquatic macrophytes release oxygen from roots into the rhizosphere and that this release influences the biogeochemical cycles in the sediments through the effects on the redox status of the sediment (see sections 2.1, 2.7.4 and 3.2.1 for more details). Oxygen release from roots may, for example, support nitrification, precipitation of Fe and Mn, dissolution of sulphides or oxidation of harmful compounds.

Root systems also release other substances besides oxygen. These substances are usually organic compounds such as anaerobic metabolites, organic acids (see section 2.4.2.4), phytometallophores (see section 2.7.4), peptides (e.g., phytochelatins), alkaloids, phenolics, terpenoids or steroids (Rovira, 1969; Barber and Martin, 1976; Neori et al., 2000). The magnitude of this release is still unclear, but reported values are generally in the range of 5-25% of the photosynthetically-fixed carbon.

Functions of the root exudates are manifold. Allelopathy, i.e., inhibition of one plant species through chemical means by another plants (Szczepanski, 1977; Rice, 1984; Putnam, 1985) has been well documented in wetlands (Gopal and Goel, 1993; Hootsmans and Blindow, 1994). Allelopathic chemicals belong to several chemical groups, particularly phenolics (Hagland and Williams, 1985), organic acids (Rao and Mikkelsen, 1977), plant hormones and plant metabolites of plant-aromatic amino acids (Gunnison and Barko, 1988), alkaloids, flavonids, terpenoids, steroids (McClure, 1970), long-chain fatty acids, and elemental sulfur (Gopal and Goel, 1993). However, it is unclear how allelopathy may affect plants in constructed wetlands.

More important for the process of wastewater treatment is the release of anti-microbial compounds by roots of wetland plants. One of the first studies reporting on the excretion of anti-bacterial substances by macrophytes was published by Drobot'ko et al. (1958). Their results showed an anti-microbial activity by alkaloids extracted from *Nuphar lutea*. Seidel (1964, 1966, 1976) showed that the *Scirpus* (= *Schoenoplectus*) *lacustris* (Bulrush) releases antibiotics from its roots and range of bacteria (coliforms, salmonella and enterococci) obviously disappeared from polluted water by passing through a vegetation of bulrushes. Vincent et al. (1994) showed the antimicrobial properties of exudates of *Mentha aquatica*, *Phragmites australis* and also for *Scirpus lacustris*. According to Gopal and Goel (1993), the substances excreted by the roots of many wetland plant species responsible for antimicrobial activity are tannic and gallic acids, however, other compounds may probably be involved as well.

Phytometallophores and phytochelatins are important in heavy metals cycling and removal in constructed wetlands. Phytometallophores (formerly phytosiderophores, see section 2.7.4) are non-proteinogenic amino-acids root exudates that chelate and mobilize Fe, Cu, Zn, and Mn (Römheld, 1991). Chelated metals then can be uptaken by roots (see section 2.7.4.) Phytochelatins may be the principal heavy-metal complexing peptides of higher plants (Grill et al., 1985; Steffens, 1990). They are metallothionein-like in function but differ in their chemical structure and composition (L-cystein, L-glutamate, and glycine at a ratio of 4:4:1) (Neori et al., 2000). The synthesis of these peptides can be induced by copper, cadmium, mercury, lead, and zinc (Salt et al., 1989; Robinson and Thurman, 1986; Fujita and Kawanishi, 1986; Fujita and Nakano, 1988). By excreting phytochelatins, plants try to limit and/or avoid heavy metal toxicity.

The macrophytes in constructed wetlands may also have functions that are not directly related to the water treatment processes. Large treatment systems may support a diverse wildlife including mammals, birds, macroinvertebrates, fish, reptiles and amphibians (Feierabend, 1989, Knight, 1996, Worall et al., 1996; Mitsch and Gosselink, 2000). The plant growing in constructed wetlands are usually not used because of the seasonality, limited quantity and unstable quality. Probably only in Africa, plants and especially *Phragmites mauritianus,* are used for various products such as mats, fences, roofs, although the quality of reeds from treatment wetlands is generally not considered high. Also, decorative plants such as *Iris pseudacorus* (Yellow flag), *Canna* spp. (Canna lilies), *Butomus umbellatus* (Flowering rush), *Filipendula ulmaria*, (Queen of the prairie) could be selected for small single-house constructed wetlands in order to provide a nice-looking part of the garden.

Chapter 4

TYPES OF CONSTRUCTED WETLANDS FOR WASTEWATER TREATMENT

Constructed wetland treatment systems are engineered systems that have been designed and constructed to utilize the natural processes involving wetland vegetation, soils, and their associated microbial assemblages to assist in treating wastewater. They are designed to take advantage of many of the processes that occur in natural wetlands, but do so within a more controlled environment. Synonymous terms to *constructed* include *man-made*, *engineered* or *artificial* wetlands.

Constructed wetlands can be built with a much greater degree of control than natural systems, thus allowing the establishment of experimental treatment facilities with well-defined composition of substrate, type of vegetation, and flow pattern. In addition, constructed wetlands offer several additional advantages compared to natural wetlands including site selection, flexibility in sizing, and most importantly, control over the hydraulic pathways and retention time (Brix, 1993a).

Hammer (1992) pointed out that unfortunately, a few of the wetland descriptions have been used synonymously and need precise definition to insure common understanding. *Restored wetlands* are areas that previously supported natural wetland ecosystems but were modified or changed, eliminating typical flora and fauna, and used for other purposes. These areas have then subsequently been altered to return to poorly drained soils and wetlands flora and fauna in order to enhance life support, flood control, recreational, educational, or other functional values. *Created wetlands* formerly had well-drained soils supporting terrestrial flora and fauna but have been deliberately modified to establish the requisite hydrological conditions producing poorly drained soils, and wetland flora and fauna in order to enhance life support, flood control, recreational, educational, or other functional values.

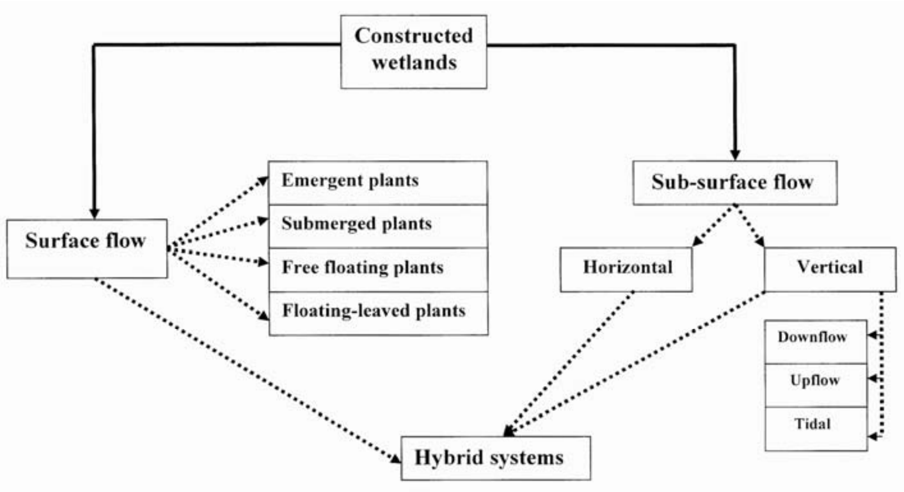


Figure 4-1. Classification of constructed wetlands for wastewater treatment.

Constructed wetlands could be classified according to the various parameters but two most important criteria are water flow regime (surface and sub-surface) and the type of macrophytic growth (Fig. 4-1). Different types of constructed wetlands may be combined with each other (so called hybrid or combined systems) in order to exploit the specific advantages of the different systems. The quality of the final effluent from the systems improves with the complexity of the facility.

4.1 Surface flow systems

Constructed wetlands with surface flow (= *free water surface constructed wetlands*, FWS CW) consist of basins or channels, with soil or another suitable medium to support the rooted vegetation (if present) and water at a relatively shallow depth flowing through the unit. The shallow water depth, low flow velocity, and presence of the plant stalks and litter regulate water flow and, especially in long, narrow channels, ensure plug-flow conditions (Reed et al., 1988). One of their primary design purposes is to contact wastewater with reactive biological surfaces (Kadlec and Knight, 1996). The FWS CWs can be classified according to the type of macrophytes (Fig. 4-1).

4.1.1 Systems with free-floating macrophytes

Constructed wetlands with free floating macrophytes consist of one or more shallow ponds in which plants float on the surface (Fig. 4-2). The shallower depth and the presence of aquatic macrophytes in place of algae are the major differences between constructed wetlands with free floating macrophytes and stabilization ponds (Kadlec et al., 2000).

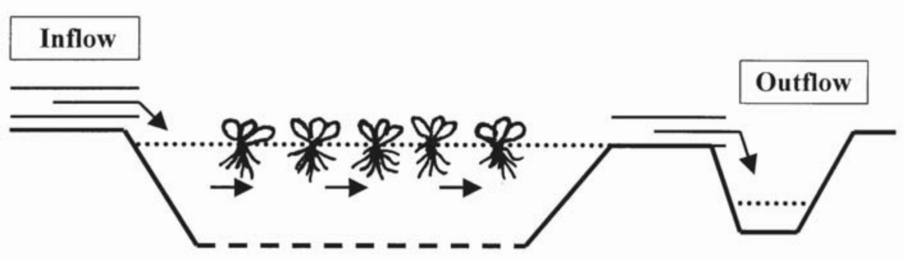


Figure 4-2. Schematic representation of the constructed wetland with free floating macrophytes. From Vymazal (2001a) with permission from Backhuys Publishers.

Free-floating macrophytes are highly diverse in form and habit, ranging from large plants with rosettes of aerial and/or floating leaves and well-developed submerged roots (e.g., *Eichhornia crassipes* - Water hyacinth or *Pistia stratiotes* - Water lettuce) to minute surface-floating plants with few or no roots (Lemnaceae - Duckweed; e.g., *Lemna* spp., *Spirodela polyrhiza*, *Wolffia* spp. (Brix and Schierup, 1989c).

Organics are principally removed by bacterial metabolism of both attached and free living bacteria. The root system of the free-floating plants provides a large surface area for attached microorganisms, thus increasing the potential for decomposition of organic matter. In order to ensure optimum contact opportunities between the wastewater and the attached microbial growth a relatively shallow reactor and a relatively low flow velocity are recommended. The mass of plants on the surface minimizes wind-induced turbulence and mixing and the removal of suspended solids occurs through gravity sedimentation in the zone under the surface layer of floating plants.

Nutrient removal in systems with free-floating plants is far more complicated that plant uptake alone. Nitrogen is removed through plant uptake (with regular harvesting), ammonia volatilization, and nitrification-denitrification. The nitrifiers can flourish attached to the roots, which provide oxygen. Nitrification also occurs in the water column when dissolved oxygen levels of the water are adequate to support activity of nitrifying bacteria. These conditions are usually created at relatively low plant densities and a partial plant cover over the water surface. As the plant density increases, O_2 diffusion into the water is restricted, thus, decreasing the dissolved O_2 levels of the water (Rai and Munshi, 1979; Reddy, 1981). Also, dense plant cover on the water surface suppresses the growth of algae growth by preventing passage of sunlight to the water column. This results in anoxic zones and created favorable conditions for nitrate denitrification which may also proceed in benthic layer if sufficient sources of organic carbon are available.

Phosphorus can be removed from these systems by microbial assimilation, precipitation with divalent and trivalent cations, or adsorption onto clays or organic matter. However, most studies have shown that plant uptake and subsequent harvest is the only reliable long-term P-removal mechanism (DeBusk and Reddy, 1987a). Harvesting is essential because detritus of plants tend to release P into water during decomposition, thus, decreasing the P removal efficiency of the system (Reddy and Sacco, 1981).

4.1.1.1 Water hyacinth

Water hyacinth, *Eichhornia crassipes* (Mart.) Solms. is one of the most prominent aquatic weed plant found throughout the tropical and subtropical areas of the world (Sculthorpe, 1967). Water hyacinth is a perennial, freshwater aquatic macrophyte with rounded, upright, shiny green leaves and spikes of lavender flowers. The petioles of the plant are spongy with many air spaces and contribute to the buoyancy of the plant. The size varies with habitat. When grown in wastewater individual plants range from 50 to 120 cm from the top of the flower to the root tips. The root length vary with the nutrient status of the water and the frequency of plant harvest (Fig. 4-3).

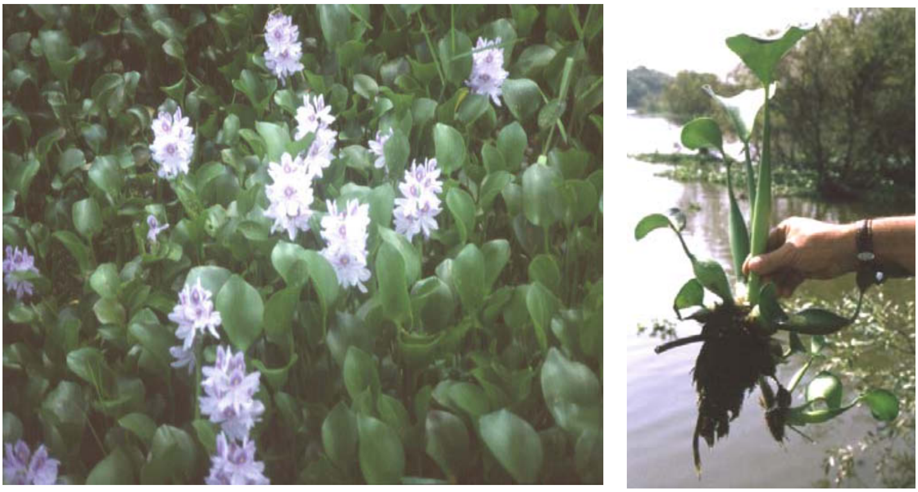


Figure 4-3. *Eichhornia crassipes* (Water hyacinth) in a natural stand in southern Florida (left) and detail (right). Photos by Jan Vymazal.

The plants spread laterally until the water surface is covered and then the vertical growth increases. Hyacinths are one of the most productive photosynthetic plants in the world. A single plant can produce approximately 65,000 offspring during a single season (Rogers and Davis, 1972) and 10 plants can completely cover 0.4 ha of a natural freshwater surface (Penfound and Earle, 1948). This rapid growth is the reason that hyacinths are a serious nuisance problem in southern waterways, but these same attributes become an advantage when used in a wastewater treatment system.

It has been shown that productivity of floating macrophytes is strongly influenced by plant standing crop, with high yields (and rapid nutrient uptake) occurring only when the plants are maintained within a certain density range (Oki et al., 1981; DeBusk et al. 1981; Reddy and DeBusk, 1984). If the standing crop exceeds the optimum range for growth (in case of Water hyacinth it is 500 – 2000 g m^{-2}) new tissue synthesis serves merely to replace senescing or decaying plant material, and no net increase in plant biomass (or nutrient uptake) occurs. Wolverton (1979) recommended wet weight plant densities from 12 to 22 kg m^{-2} (about 600-1000 g m^{-2} dry mass) for optimum treatment with loosely packed plants with 80 to 100% surface coverage. Similar values are also reported by DeBusk and Reddy (1987b); the authors found the maximum productivity at plant biomass density about 1 kg dry mass m^{-2}. Fisher and Reddy (1987) compared P and N uptake by water hyacinths in harvested and unharvested channels and found that in harvested channels the uptake was higher by 6% for nitrogen and by 12% for phosphorus.

Phosphorus uptake by Water hyacinth is in direct proportion to N availability (Haller and Sutton, 1973; Reddy and Tucker, 1983), suggesting that P use efficiency by water hyacinth does not depend only on the P concentration in the water but also on the N:P ratio of the water (Reddy et al.,1989). Other studies revealed that substantial amounts of nitrogen are removed by water hyacinth, but removal of P is slow and affected by N/P ratio (Ornes and Sutton, 1975; Boyd, 1976; Mitsch, 1977).

Since the late 1960s the use of Water hyacinth for treatment of various types of wastewater has been tested in small experimental systems, in countries where this plant naturally occurred, namely in India, southeastern Asia, southern Japan and southern parts of the United States. The types of wastewater included sugar refinery (Sinha and Sinha, 1969, Yeoh, 1983), dairy (Aowal and Singh, 1982; Trivedy et al., 1983; Goel et al., 1985), distillery (Trivedy and Khomane, 1985), piggery (Yeah, 1983), palm oil (Yeoh, 1983; John, 1983, 1984), natural rubber (John, 1984), tannery (Trivedy et al., 1983; Haider et al., 1983), textile (Widyanto, 1975; Trivedy and Khomane, 1985), electroplating (Shroff, 1982; Yeoh, 1983), heavy metals (Jayaraman and Prabhakar, 1982; Das, 1984; Yeoh, 1983; Tatsuyama et al., 1977; Muramoto and Oki, 1983, 1984), pesticides (Gudekar et al., 1984), phenol (Wolverton and McKown, 1976; O'Keeffe et al., 1987; Vaidyanathan et al., 1983;) fluoride (Rao et al., 1973).

The experiments aimed at the uptake of phosphorus and nitrogen by water hyacinth were quite optimistic. The removal rates for nitrogen in natural stands, wastewater effluents and nutrient cultures varied between 48 and 73 g N m^{-2} yr^{-1} (Para and Hortensine, 1974; Reddy et al., 1982), 120 and 173 g N m^{-2} yr^{-1} (Boyd, 1976; Reddy et al., 1983) and 357 to 535 g N m^{-2} yr^{-1} (Sato and Kondo, 1981; Reddy and Tucker, 1983; Reddy et al., 1989), respectively. The respective values for phosphorus were 9.3 – 16 g P m^{-2} yr^{-1}, 14 – 39 g P m^{-2} yr^{-1} and 99 – 126 g P m^{-2} yr^{-1}. Steward (1970) estimated the removal of nitrogen and phosphorus up to 600 g N m^{-2} yr^{-1} and 60 g P m^{-2} yr^{-1}.

Despite many experimental efforts, only limited number of full-scale Water hyacinth-based constructed were built with majority being built during the 1980s and early 1990s in southern states of the United States (Fig. 4-4). Treatment effect is usually very high for organics and suspended solids and removal of nutrients is strongly dependent on harvesting management as the major pathway of nutrient removal is uptake by plants. (Reed et al., 1988; Bastian and Reed; 1979, Reed and Bastian, 1980). Treatment effect for various Water hyacinth based systems is presented in Table 4-1. Removed amounts of nitrogen reached the values obtained previously in experiments with nutrient cultures while removed amounts of phosphorus were somewhat lower.

Figure 4-4. Constructed wetland with Water hyacinth Aqua III in San Diego, California, USA. Photo by George Tchobanoglous, with permission.

Table 4-1. Treatment efficiency of full scale constructed wetlands with Water hyacinth. Results from 17 systems*.

Parameter	**Concentration**			**Loading**		
	Inflow	**Outflow**	**Efficiency**	**Inflow**	**Outflow**	**Removal**
	(mg l^{-1})		**(%)**	**(kg ha^{-1} d^{-1})**		
BOD_5	80	14.1	75.7	96	20	76
TSS	48.2	9.3	64.4	109	56	53
				(g m^{-2} yr^{-1})		
TN	14.6	6.6	59.5	838	431	407
TP	3.8	2.2	46.7	200	127	73

*Japan (Oki, 1983), USA-Texas (Dinges, 1979; Dinges and Doersam, 1987), Iowa (Wooten and Dodd, 1976), Mississippi (Wolverton and McDonald, 1976, 1979), Florida (Kruzic, 1979; Stewart, 1979; Swett, 1979; Stewart et al., 1987; Reddy et al., 1982), California (Crites and Tchobanoglous, 1998), Thailand (Orth and Sapkota, 1988).

The two major reasons for the fact that constructed wetlands with water hyacinth have not spread more are high operation and maintenance costs and, of course, limited growth under temperate and cold climatic conditions. A frequent plant harvest is necessary to sustain high levels of nutrient removal. With intensive harvesting, it is necessary to construct the hyacinth ponds so that harvesting can be easily accomplished. This has a tendency to increase the cost of the hyacinth system, and also develops the problem of disposing the excess material (Middlebrooks, 1980). Harvest techniques for aquatic macrophytes were reviewed by Bagnall (1986). As the biomass is voluminous, heavy equipment such as cranes and conveyors are used for the harvest. Also, the further use of the biomass is questionable.

Since the hyacinth plants are about 95% water, an intermediate drying step is usually employed prior to disposal or utilization of the harvested material at the smaller systems. The dried plants can be disposed of in a landfill, or elsewhere, as permitted by the local regulatory authorities. If the wastewater has very high metal concentrations it may be advisable to check the metal content of the dried plant to ensure that the levels do not exceed permit allowances for disposal or utilization. The simplest approach for beneficial reuse of the harvested materials is to compost the semi-dry hyacinths and then use the material as a soil conditioner or fertilizer. Anaerobic digestion of the plants and sludge for methane production and processing of the plants for animal feed have been shown to be technically feasible but marginally cost effective (Reed et al., 1988).

Treatment systems with *Eichhornia crassipes* were sufficiently developed to be successfully applied in the tropics and subtropics. The major reason for the limited use in temperate regions or regions with even colder climatic conditions is the fact that Water hyacinth is strongly damaged by frost, and the growth rate is greatly reduced at temperatures below 10 °C (Reddy and Bagnall 1981, see also Fig. 4-5). Žáková et al. (1994) found out that the application of water hyacinth systems for wastewater treatment and nutrient removal under northern temperate climatic conditions of the Czech Republic is realistic but only for a very short period of time. The optimum water temperature for Water hyacinth growth (> 25 °C) was reached only for a short period of time from the end of June to the beginning of July and it also was found that growth appeared only when water temperature was ≥ 18 °C. The strong influence of temperature on phosphorus uptake by *Eichhornia crassipes* was also reported by Urbanc-Berčič and Gaberščik (1989) from Slovenia. Dobelmann et al. (2000) reported successful use of Water hyacinth for wastewater treatment over a 125-day period in Baden-Baden, Germany.

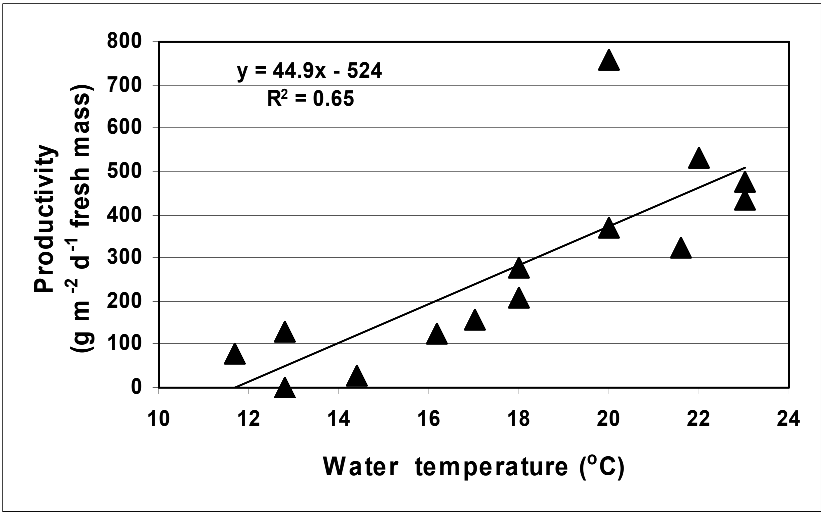


Figure 4-5. Productivity of water hyacinth (*Eichhornia crassipes*) at constructed wetland Aqua III in San Diego, California, for the period of August 1994 through September 1995. The points depict an average water temperature for each month. Data from Crites and Tchobanoglous (1998).

There was an explosion of research studies on the use of Water hyacinth for wastewater treatment in the 1970s and early 1980s. However, after this period very few information appeared in the scientific literature mostly because these systems are not economic. Stewart et al. (1987) pointed out that while the concept of using Water hyacinth for wastewater treatment had been extensively investigated and reviewed for nearly 20 years, there still remains little information regarding the operational needs associated with large scale applications. In Table 4-2, the major design parameters for constructed wetlands with Water hyacinth are summarized. However, it seems that these criteria are rather theoretical because at present, there are very few operational systems where these criteria could be evaluated.

Table 4-2. Design criteria for Water hyacinth wastewater treatment systems operated in warm climates (Middlebrooks, 1980; Tchobanoglous, 1987; Reed et al., 1988; Crites and Tchobanoglous, 1998).

Parameter	**Level of treatment**			
	Primary	Secondary Aerated	Secondary Non-aerated	Tertiary
Hydraulic loading rate (cm d^{-1})	2	15 - 28	2.8 – 6.5	< 8
Hydraulic retention time (d)	> 40	4 - 8	10 - 30	≤ 6
Water depth (m)	< 1.5	1.2 – 1.4	0.45 – 0.75	<< 0.9
Org. load. rate (kg BOD_5 ha^{-1} d^{-1})	50	280 - 500	70 - 90	< 50
Max. area of a single cell (m^2)	4000	4000	4000	4000
Length:width ratio	> 3:1	> 3:1	> 3:1	> 3:1
Water temperature (°C)	> 10	> 10*	> 10*	> 20
Harvest schedule**	A to S	M to W	A to S	MP-EFW

*preferably > 20°C; **A = annually, S = seasonally, M = monthly, W = weekly, MP-EFW = mature plants, every few weeks

Constructed wetlands with Water hyacinth have been revived recently and attempts have been made to use these systems again. Maine et al. (2006) reported the use Water hyacinth system for the treatment of wastewaters from a tool factory in Santo Tomé, Argentina. The 2,000 m^2 system was quite effective in removing BOD_5 (inflow 195 mg l^{-1}, outflow 38 mg l^{-1}), nitrate (inflow 16.8 mg l^{-1}, outflow 3.2 mg l^{-1}), iron (inflow 12.2 mg l^{-1}, outflow 0.31 mg l^{-1}) and chromium (inflow 12 mg l^{-1}, outflow 2.3 mg l^{-1}). Tua et al. (2006) reported the use of Water hyacinth-based constructed wetland for the treatment of fish processing wastewater in Vietnam. Perdomo et al. (2000) reported experimental use of Water hyacinth for dairy wastewaters treatment in Uruguay. The system performed well but parallel system with *Typha* spp. exhibited better removal effect. Elias et al. (2001) used Water hyacinth pond as a part of treatment system for public water supply in Brazil. Kalibbala et al. (2002) used ponds with *Eichhornia* for tertiary treatment of brewery wastewater in Uganda. The removal efficiency was not high but similar to *Cyperus papyrus* (grown in floating mode). However, the experiments proved that Water hyacinth can grow in water with $BOD_5 > 900$ mg l^{-1}.

4.1.1.2 Duckweed

Duckweed (Lemaceae) are tiny free-floating vascular plants (Fig 4-6) with world-wide distribution. Their communities occur, as a rule, in small water bodies, namely in fishponds, ditches or lagoons (Fig. 4-6). The family (Lemnaceae) consists of about 35 species in four genera, *Lemna* (e.g., *Lemna minor* L., *Lemna trisulca* L.), *Spirodela* (*Spirodela polyrhiza* (L.) Schleiden), *Wolffiella* and *Wolffia* (Hillman, 1961; Landolt, 1980a, b). Taxonomically the family is complicated by clonal characteristics. Thus, stands of the same species from various locations can differ in a variety of morphological and physiological characters (Landolt, 1957; Rejmánková, 1975; Porath et al., 1979).

Duckweed are among the fastest growing plants in the world, frequently doubling their biomass under optimal conditions in 2 or 3 days (Rejmánková, 1971; Said et al., 1979; Rusoff et al., 1980) with night growth equaling or exceeding day growth for some species (Mestayer, 1980). Duckweed, compared to Water hyacinth, have a much wider geographic range as they are able to grow at temperatures as low as 1 to 3 ^{o}C (Brix 1993a). Culley and Epps (1973) pointed out that duckweed will survive light freezes and even continue to grow during the winter months. Duckweed growth optimum is at temperatures between 20 and 30 ^{o}C, but serious effects will occur at temperatures between 35 and 40 ^{o}C. Below 17 ^{o}C some duckweeds show a decreasing rate of growth. A full, thick mat of duckweed plants may have a temperature about 10 ^{o}C above the ambient air because of radiation effects (Stephenson et al., 1980; Reed et al., 1988).

Fig. 4-6. Duckweed (Lemnaceae). Left: detail, right: pond covered with Duckweed. Photos by Jan Vymazal.

Duckweed can tolerate a wide range of pH (3.0-10.0), but displays optimum growth in a medium of 5.0 – 7.0 (Stephenson et al.,1980). A pH value greater than 10 will have serious effects on growth. A diurnal pH of 10 is possible in ponds only partially-covered with duckweed because of algal activity. At high pH values duckweed growth is inhibited, and the mat may never expand to cover the entire pond. The plants can grow in full sunlight as well as dense shade. *Wolffia* does better under darker conditions, *Lemna gibba* does better in sunlight (Zirschky and Reed, 1988).

Duckweed, compared to Water hyacinth, play a less direct role in the treatment process as they lack extensive root systems and therefore provide a smaller surface area for attached microbial growth. A dense cover of Duckweed on the surface of water inhibits both oxygen entering the water by diffusion and the photosynthetic production of oxygen by phytoplankton because of the poor light penetration. The water becomes largely anoxic, which in turn favors denitrification (Brix 1993a). The surface mat reduces light penetration by 35% and 94% at standing crops of 500 and 3900 g m^{-2} (Ngo, 1987).

The growing plants will first form a single layer covering the entire water surface. Then, in some species, additional plants will grow on top of each other resulting in a mat 1 to 2 cm thick (Zirschky and Reed 1988). The small size of duckweed makes the plants susceptible to the wind and it may result in part of the basin being uncovered so floating booms or cells are frequently used to hold the plants in place (Fig. 4-7).

Figure 4-7. Duckweed is susceptible to wind in natural systems (left, photo by Jan Vymazal) and therefore it is useful to use floating cells to keep the plants in place and keep the whole surface covered (right). Right: harvest of *Lemna* system (with permission).

Nitrogen can be removed by direct plant uptake with frequent biomass harvesting, ammonia volatilization and microbial nitrification-denitrification (Reed et al., 1988; Bonomo et al., 1996). Nitrification of ammonia occurs within the aerobic root zone of duckweed. However, this layer is very thin and therefore little nitrification occurs in duckweed-based systems. Denitrification occurs in the much larger reduced zones in the water column. However, the nitrification/denitrification complex is not very effective for nitrogen removal because of limited nitrification in the system. Bonomo et al. (1996) pointed out that nitrogen removal can only be reasonable in duckweed ponds with the use of supplemental aeration in order to enhance nitrification or in ponds with already nitrified wastewater. Volatilization may contribute substantially to the nitrogen removal in duckweed-based treatment wetlands. Billore et al. (1994) reported an average ammonia loss from a sewage-fed system in India 125 mg $m^{-2}d^{-1}$ with maximum values 269 mg m^{-2}. Interestingly, ammonia loss from a duckweed system was higher than from the system with no plants (average loss of 41 mg m^{-2}). According to the results of laboratory experiments, duckweeds tolerate as high concentration of elemental nitrogen as 375 mg l^{-1} (Rejmánková, 1979).

The major route for phosphorus removal is direct plant uptake. Other removal mechanisms play negligible role in phosphorus removal (Sutton and Ornes, 1975). Culley et al. (1978, 1981) reported good growth of duckweed within the P concentrations of 6 to 154 mg l^{-1}. On the other hand, reduced growth in some species occurs only after P values dropped below 0.017 mg l^{-1} (Lüönd, 1980).

Duckweed can accumulate large amounts of mineral nutrients in their tissues. Hillman and Culley (1973) reported that as for inorganic materials, duckweed absorb the essential plant macronutrients at rates and in proportions that obviously depend on the concentrations of nutrients and the rate of harvest, if any, as well as on whether any simple nutrient becomes limiting long before any other. Rejmánková (1979) reported concentrations as high as 7% N in dry mass of *Lemna gibba* and 2.9% P in the dry mass of *Spirodela polyrhiza* obtained under laboratory conditions. When growing in wastewater concentrations of N may reach the values up to 7.3% (Mbagwu and Adeniji (1988) and concentration of P may exceed 2.5 % (Culley and Epps 1973; Culley et al., 1981). However, concentrations higher than 4% N and 1% P on dry mass basis are commonly found in natural wetlands (Allenby, 1968; Dykyjová and Květ, 1982; Rejmánková, 1978; Rejmánková and Hapala, 1982). Uptake rates are comparable with those reported for water hyacinth. Ngo (1987) reported the uptake rate of 611 g N m^{-2} yr^{-1} and 80 g P m^{-2} yr^{-1} while Květ et al. (1979) estimated 73 g N m^{-2} yr^{-1} for temperate conditions in the Czech Republic. Culley et al. (1981) calculated mean annual nutrient uptake by mixed culture of duckweeds (*Spirodela polyrhiza, S. punctata*, *Lemna gibba*, *Wolffia columbiana* in approximate equal amounts) as 138 g N m^{-2} yr^{-1} and 34.5 g P m^{-2} yr^{-1}.

Rejmánková (1982) pointed out that harvesting of duckweed is essential in order to maintain high growth rate (and nutrient uptake). In addition, if duckweed is not removed from the pond, the nutrients are very rapidly released from their decomposing detritus back into the water. Said et al. (1979) reported that in an outdoor tank culture system with fresh dairy cattle manure maximum growth of *Spirodela polyrhiza* was obtained when harvested at 6-day intervals. The data show that daily harvest and maintenance of a near optimum standing stock yields 58% more duckweed than harvesting once in 6 days. The authors pointed out that optimum standing stock for maximum yield by daily harvest is close to 550 g fresh duckweed m^{-2}. Rejmánková et al. (1990) reported that under a given intrinsic growth rate determined by climatic conditions and non-limiting mineral resources, the yield depends on the initial biomass and on both the size and frequency of harvest. The optimum harvesting regime can be assessed in two ways (Rejmánková et al., 1990): 1) empirically, as a choice of the best of many variants tested and 2) mathematically, if the growth function is known and estimates of its parameters can be made. In the majority of cases, the studies on duckweed harvesting were exclusively empirical.

Valderrama and Ahumada (2006) evaluated two harvesting strategies that are usually applied to natural populations – constant quota (CQ) and constant effort (CE). With CQ, a fixed amount of the biomass is harvested with a given frequency, independently of the population density. Maximum yields are obtained when the amount harvested at each time is equal to the maximum production of the population (May et al., 1979). With CE the harvested biomass is not a fixed amount but a fixed proportion of the population, which in natural populations could mean a fixed number of hours of harvesting or a fixed number of harvesters with the same efficiency. Although CE does not yield always the same amount of biomass, it can be more sustainable for the population in the long run (Braumann, 2001). The authors reported that the model predicted well population dynamics and biomass production with the both strategies. CE method was found to be a more suitable strategy since: a) it allowed a sustainable harvesting, b) it maintained a complete cover of the reactor's surface and c) could yield a higher biomass production in the long-term.

DeBusk et al. (1981) assessed the optimum stocking density as 38 g m^{-2} and then designed their sampling so as to keep this density more or less constant throughout the whole experiment. Another approach used by Reddy and DeBusk (1985) was to start with low density (10 g m^{-2}), allow plants to grow until maximum densities were reached and then harvest and restock to original low densities. Rejmánková et al. (1990) evaluated the optimum harvest fraction as a function of the time interval between successive harvests, the biomass at the start of each interval, and expected intrinsic growth rate but the highest experimental yield obtained at 25% harvest fraction fell short of the theoretically obtainable maximum yield by 27.2%.

The nutritive value of the produced Duckweed biomass is high compared to that of the water hyacinth, as it contain at least twice as much protein, fat, nitrogen, and phosphorus (Culley and Epps, 1973; Hillman and Culley, 1973). The crude protein content of duckweeds obtained from natural waters has been reported to range from 7 to 20% (Bhanthumnavin and McGarry, 1971; Rusoff et al., 1980) but if grown in enriched waters such as agricultural or municipal waste lagoons, the protein content of Duckweed (30-40%) is greatly increased over that from natural waters (Culley and Epps, 1973; Hillman and Culley, 1973; Russoff et al., 1980). Further, the plants growth response is better to ammonia than to nitrate or nitrite (Ferguson and Bollard, 1969). It has been suggested that dried duckweed may be used as a fertilizer (e.g., So 1987) or animal feed (Truax et al., 1972; Culley and Epps, 1973; Harvey and Fox, 1973; Culley et al., 1981). However, the field results indicate that the mass of harvested plants represents an unfavorable ballast that is difficult to utilize (Ozimek, 1996).

The use of duckweed for wastewater treatment is not as well developed as hyacinth systems even though that first attempt to use these plants were carried out 30 years ago (Vymazal, 1998). The retention time in duckweed-based wastewater treatment system depends on wastewater quality, effluent requirements, harvesting rate, and climate, but it varies typically from 10 to 30 days. A screen or other baffling system is essential at the outlet of the basin to prevent loss of the small floating plants with the effluent (Reed et al., 1988). The system depth varies between 0.4 and 3.0 m (Zirschky and Reed, 1988; Hancock and Buddhavarapu, 1993; Crites and Tchobanoglous, 1998). Organic loading for Duckweed systems is recommended up to about 22 – 34 kg BOD_5 ha^{-1} d^{-1}. Hydraulic loading rate is recommended < 5.2 cm d^{-1} (Crites and Tchobanoglous, 1998).

Removal of organics and suspended solids in Duckweed-based constructed wetlands is presented in Table 4-3. The outflow concentrations are quite low but organic loading of these systems is usually quite low. Crites and Tchobanoglous (1998) pointed out that The Lemna Corporation suggests that wastewater entering the Duckweed system should be partially treated to a BOD_5 level of 60 mg l^{-1} or less by facultative ponds, aerated ponds or mechanical treatment plants. To achieve a 20 mg l^{-1} BOD_5 in the effluent, Lemna Corp. suggest a target influent of 40 mg l^{-1}, 20-day detention time, and a pond sizing of 12.8 m^2 m^{-3} d^{-1}, i.e. 19.2 m^2 PE^{-1}. To achieve a final BOD_5 of 10 mg l^{-1} out of influent of 30 mg l^{-1} it is suggested to use about 37 m^2 PE^{-1}. That is about eight times more than the average size of horizontal sub-surface flow constructed wetlands.

Table 4-3. Typical effluent concentrations of BOD_5 and TSS from constructed wetlands with Duckweed. Data from Zirschky and Reed (1988) and Crites and Tchobanoglous (1998).

Location	**BOD_5 (mg l^{-1})**	**TSS (mg l^{-1})**	**Detention Time (d)**
Mamou, Louisiana	5	8	30
White House, Tennessee	3	4	27
Four Corners, Louisiana	3	3	24
Biloxi, Mississippi	15	12	21
Ellaville, Georgia	13	10	20
Kyle, Texas	18	13	12
Nokesville, Virginia	6	5	12
Arkadelphia, Arkansas	12	15	10
NSTL, Mississippi	3	11.5	8
Collins, Mississippi	13	13	7
Paragould, Arkansas	49	42	6
Wilton, Arkansas	6.5	7.4	0.7

There is a lack of information about the treatment efficiency of operational Duckweed-based systems for nutrient removal from wastewaters. Buddhavarapu and Hancock (1991) reported that total P was removed up to 89% in the pilot-scale demonstration project in North Dakota and duckweed harvest accounted for up to 73% of the removed phosphorus. Combining phosphorus content in harvested and unharvested duckweed resulted in a potential phosphorus uptake as high as 91% with outflow concentrations of TP less than 0.15 mg l^{-1}. Cadelli et al. (1998) reported the use of *Lemna* ponds in series behind aerated stabilization pond in two sites in Belgium. The average removal of total-N and total-P in these systems was 46 and 48%, respectively. Alaerts and Al-Nozaly (2000) reported removal of up to 365 g N m^{-2} yr^{-1} and 119 g P m^{-2} yr^{-1} in a pilot plant in Yemen. Those values are within the range found under experimental conditions (see previous text). Andreottola et al. (1994) reported that to achieve 75% removal of nitrogen requires 23-78 m^2 PE^{-1} considering 1 PE = 10 g N d^{-1}.

Mdano (2002) reported that both *Lemna* and *Spirodela* survived and multiplied greatly in pulp mill wastewater in Tanzania with COD concentrations between 60 and 400 mg l^{-1} and still survived and multiplied up to COD concentration of 1,525 mg l^{-1}. However, there was hardly any growth of Duckweed in paper mill wastewater without nutrients additions. On the other hand, Kalibbala et al. (2002) reported that Duckweed was not able to grow in brewery wastewater with BOD_5 > 900 mg l^{-1} in Uganda.

The results from Poland, where about 35 Lemna Corporation systems are in operation (Ozimek, 2003; Ozimek and Czupryňski, 2003) indicate quite disappointing performance. Only 0.2% of nitrogen and 0.4% of phosphorus flowing annually into “Lemna system” in Mniów can be removed with harvesting of plants. The main problems associated with the use of the Lemna systems are the variability of plant biomass and cover from year to year, and the lack of any information on what affects the variability in the plant cover. The research indicated that Duckweed ponds do not improve the efficiency of wastewater treatment and they do generate biomass with no other use (Ozimek and Czupryňski, 2003).

4.1.1.3 Other free-floating macrophytes

The use of free-floating plants other than water hyacinth and duckweed is quite limited and not well documented in the literature. Other free-floating plants used for wastewater treatment include mostly *Pistia stratiotes* L. (Water lettuce) and *Ipomoea aquatica* Forsh. (Water spinach). Laboratory experiments have been also done with *Hydrocotyle umbellata* L. (Pennywort).

Pistia stratiotes (Fig. 4-8) is spongy and consists of a shell-like rosette of light yellow-green leaves. A tuft of long unbranched fibrous roots extends from a central extension of the underwater stolon. Reproduction is predominantly by budding either from the main plant or from stolons. Water lettuce appears to thrive best in still water or areas of nominal flow. It grows mainly as free-floating plant, but can survive as a semi-rooted plant for prolonged periods (Tarver et al., 1988). *Pistia* is frost sensitive and does not thrive in temperate zones. Therefore, its use in the process of wastewater treatment is limited to tropical and subtropical regions.

Ipomea aquatica (Fig. 4-9) is native to southeast Asia, Taiwan and southern China. It is found throughout the tropical regions around the world. Water spinach, a herbaceous perennial or occasionally an annual aquatic vine, with long hollow stems that trail over mud or float on the water surface. Vegetative reproduction, by the formation of lateral branches at the nodes, is the primary means of reproduction. It occurs in flooded lowland areas, wetlands, roadside ditches, rice fields, muddy riverbanks, and along or floating in stagnant streams and lakes. It prefers soggy soils and cannot survive in areas which receive frost and snow (Tarver et al., 1988).

Figure 4-8. Pistia stratiotes (Water lettuce) - detail (left) and in a natural stand in the south Florida (right). Photos by Jan Vymazal.

Figure 4-9. Ipomea aquatica (Water spinach). Left: detail from Tarver et al. (1988), right: in natural stand, courtesy of the Florida Department of Environmental Protection.

Hydrocotyle umbellata (Fig. 4-10) grows up to 30 cm tall, new leaves arise singly from each node on slender creeping stems. It is found rooted on mud along pond and canal margins and grows well on the surface of water as a floating mat.

Figure 4-10. Hydrocotyle umbellata (Pennywort). Left: *Hydrocotyle* growing in a facultative pretreatment lagoon in a constructed wetland San Jacinto, Uruguay. Right: detail. Photos by Jan Vymazal.

Pennywort was found to be an effective counterpart to Water hyacinth for winter season P removal (Clough et al., 1987) as P uptake by Pennywort was approximately the same during both summer and winter seasons - 86 and 81 mg P m^{-2} d^{-1}, respectively (DeBusk and Reddy, 1987a). It has been suggested that Water hyacinths and Pennywort can be alternately cultured, winter and summer, in order to maintain the performance at a high level year-round (Reddy and DeBusk, 1984; Wolverton and McCaleb, 1987). On the other hand, *Pistia* does not thrive in cold weather. DeBusk and Reddy (1987a) reported that nitrogen and phosphorus uptake rates by *Pistia* dropped sharply nearly 70% (in case of P from 218 to 72 mg m^{-2} d^{-1}) during the winter as compared to summer period.

Hydrocotyle and *Pistia* have demonstrated considerable potential for nutrient removal from wastewater due to their rapid growth and high nutrient assimilative capacity. As the turnover time for these plants is relatively brief, they can provide only short-term nutrient storage (Reddy and DeBusk, 1987). The experimental system with *Hydrocotyle* exhibited similar nitrogen and phosphorus removal rates (0.81 g N m^2 d^{-1} and 0.20 g P m^{-2} d^{-1}) as that planted with *Sagittaria* rates (0.81 g N m^2 d^{-1} and 0.23 g P m^{-2} d^{-1}) and higher than beds planted with *Scirpus* (0.50 g N m^2 d^{-1} and 0.12 g P m^{-2} d^{-1}) or unplanted gravel bed rates (0.11 g N m^2 d^{-1} and 0.05 g P m^{-2} d^{-1}).

Both Pennywort and Water lettuce provide a high O_2 transport (Moorhead and Reddy, 1988, 1990; Perdomo et al., 2005) and provides high growth rate and a high nutrient uptake capacity, even during relatively cold periods in subtropical regions. DeBusk et al. (1987) reported that Pennywort exhibited by far higher oxygen transport rates (112 mg O_2 g^{-1}(root) d^{-1}) as compared to *Sagittaria latifolia* (19 mg O_2 g^{-1}(root) d^{-1}) and *Scirpus americanus* (4 mg O_2 g^{-1}(root) d^{-1}). DeBusk and Reddy (1987a) reported that optimum standing crop for Pennywort ranges between 250 and 650 g m^{-2}; higher biomass does not enable high nutrient uptake rates. The optimum range for Water lettuce is 200-700 g m^{-2}. Wolverton and McCaleb (1987) reported the use of combined Pennywort/Duckweed system for wastewater treatment. The system performed even better than Water hyacinth for BOD_5 removal.

DeBusk et al. (1990) compared the efficiency of gravel-based sub-surface units planted with emergent species and units with floating *Hydrocotyle umbaellata* during the period autumn 1987 and winter 1990 in south Florida (Table 4-4). Units with *Hydrocotyle* exhibited slightly lower efficiency as compared to gravel beds either planted with *Sagittaria latifolia* (Common arrowhead, Wapato) and *Scirpus pungens* (Common threesquare) or unplanted. However, the removal effect oh *Hydrocotyle*-based units was quite high. Removal efficiency for nitrogen and phosphorus was similar for all plants used and slightly higher than unvegetated units.

Table 4-4. Treatment performance of experimental units for domestic wastewater in south Florida. Data from DeBusk et al. (1990).

	Inflow	**Outflow (mg l^{-1})**			
	(mg l^{-1})	***Hydrocotyle***	***Sagittaria***	***Scirpus***	**Gravel only**
BOD_5	177	34	19	18	22
TSS	50	6	4	4	4
	Inflow load	**Removed load**			
	(kg ha^{-1} d^{-1})	**(kg ha^{-1} d^{-1})**			
BOD_5	180	141	157	158	155
TSS	51	44	46	46	47

Agendia et al. (1996) reported the results from a *Pistia*-based sewage treatment plant (total ponds surface area of 702 m^2) serving 650 people from Cameroon (Table 4-5). The efficiency of the system was linked to the management of *Pistia* as it grows vigorously doubling its biomass in just over 5 days and tripled in 10 days. Recommended harvesting of part of the biomass was set at every 25 days – during this time the biomass increased 8 times and from 30 plants originally cultivated, 487 new ones were counted in 25 days. Harvested biomass can be used for composts and animal feed. Some nuisance as emanation of odor, mosquito and fly proliferation were associated with the system. Kalibbala et al. (2002) reported that *P. stratiotes* was not able to grow in brewery wastewater with BOD_5 > 900 mg l^{-1} in Uganda.

Kao et al. (2001) used *Pistia*-based constructed wetland for treatment of wastewaters from a university campus in Kaohsiung in southern Taiwan. *Pistia*-planted wetlands were the first part of two-stage system where the second part was a sub-surface flow wetland planted with *Phragmites australis*. Based on results from the pilot-scale study a field-scale constructed wetland (1 200 m^2) was built inside the university campus for approximately 85 m^3 d^{-1} of untreated water from the school drainage pipes and stormwater runoff. The wetland was able to reduce 60% of TSS, >85% N, 85% of Zn and Pb and 81% of COD.

Table 4-5. Treatment performance of *Pistia*-based constructed wetland at Biyeme Assi, Cameroon. From Agendia et al. (1996) with permission of Institute for Water Provision, Water Ecology and Waste Management, Universität für Bodenkultur, Vienna.

Parameter	**Inflow ($mg\ l^{-1}$)**	**Outflow ($mg\ l^{-1}$)**	**Efficiency (%)**
BOD_5	605	85	86
COD	1,780	195	89
TSS	521	25	95
NO_3-N	57	5.7	90
NH_4-N	70	43	39
PO_4-P	15.5	4.05	74
SO_4^{2-}	183	20	89
Dissolved O_2	075	6.02	88

Karnchanawong and Sanjitt (1994) reported the use of Water spinach for water treatment in pilot-scale ponds in Thailand. The authors pointed out that treatment effect was much better as compared to parallel facultative pond. Removal of total nitrogen was high at low inflow loadings, however, decreased with increasing load rates (Table 4-6).

Table 4-6. Treatment performance of Water spinach (*Ipomea aquatica*) ponds in Thailand (data from Karnchanawong and Sanjitt, 1994).

	In	**Out**	**Removal**	**In**	**Out**	**Removal**
	Org. loading 15.9 kg BOD_5 ha^{-1} d^{-1}			**Org. loading 159 kg BOD_5 ha^{-1} d^{-1}**		
BOD_5	25.4	7.1	72	25.4	14.1	45
COD	66.4	21.6	68	66.4	40.3	39
SS	14.4	1.4	90	14.5	8.0	45
NH_3-N	16.7	3,6	79	16.7	17.1	-3
TN	20.0	4.9	76	20.0	19.2	4

4.1.2 Systems with floating-leaved macrophytes

Floating-leaved macrophytes include plant species which are rooted in the substrate and their leaves float on the water surface. *Nymphaea* spp. (waterlilies, Fig. 4-11), *Nuphar lutea* (Yellow waterlily, Spatterdock, Fig. 4-11) and *Nelumbo nucifera* (Indian lotus) are typical representatives of this group.

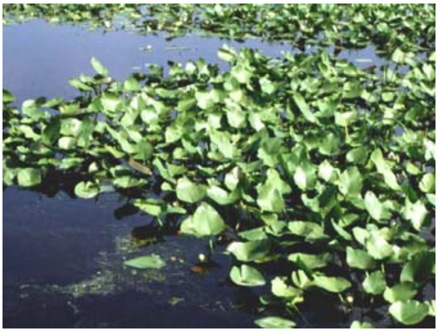


Figure 4-11. *Nymphaea odorata* Aiton (left) and *Nuphar lutea* (L.) Sibth. and Smith (right) are typical representatives of rooted plants with floating leaves. Photos by Jan Vymazal.

The schematic representation of a constructed wetland with floating-leaved macrophytes is in Figure 4-10. In fact, this type combines the features of constructed wetlands with other types of macrophytes. The plants have huge underground organs (see Fig. 3-6), leaf peduncles grow through the water column and leaves float on the water surface. In some species, such as *Nuphar*, the leaves may actually be partially above the water surface. However, peduncle density is much lower as compared to submerged or emergent species and leaves rarely cover the entire surface of the pond.

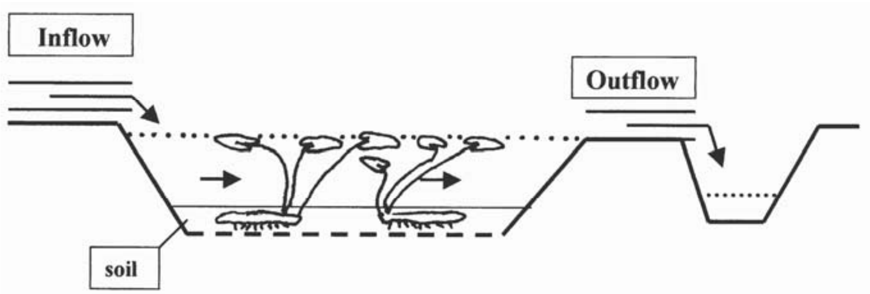


Figure 4-12. Schematic representation of a constructed wetland with floating-leaved macrophytes. From Vymazal (2001a) with permission from Backhuys Publishers.

Constructed wetlands with floating leaved macrophytes have a good potential for removal of suspended solids as leaves on the surface minimize the effect of wind causing potential resuspension and water movement. Organics are predominantly removed via sedimentation of particulate matter and microbial degradation. There are very limited data on these systems, however, it is likely that these processes are mostly aerobic in the water column due to oxygen production by algae.

In the water column, nitrogen removal may be accomplished through volatilization at higher pH values created by photosynthesis of plankton and periphytic community attached to the leaf peduncles and blades. Occurrence of algae in these systems is very likely as leaves of floating-leaved species very often do not completely cover the water surface. Aerobic conditions may favor nitrification while denitrification is unlikely in the water column but it may appear in the sediment-water interface in case of anoxic conditions. Phosphorus can be taken up in the water column by algae but this removal compartment is only short-term and nutrients are washed out quickly from decaying algal tissue. Both nitrogen and phosphorus can be taken up by roots and rhizomes. Twilley et al. (1977) showed that phosphorus could be absorbed also by submerged and floating leaves of *Nuphar lutea*. However, this absorption is 10-20 times lower as compared to root adsorption depending on the season. The highest phosphorus absorption by leaves was observed during summer.

So far, only a few systems have used this type of vegetation, e.g. at Bainikeng in China (Fig. 4-13), one pond is planted with *Nelumbo nucifera* together with Water hyacinth (Wang et al. 1994, Yang et al. 1994). It is difficult to assess the effect of *Nelumbo* because it was mixed with *Eichhornia* but the overall effect of this pond was very low (Table 4-7). Low treatment effect is, for sure, influenced by extremely high hydraulic loading rate of nearly 100 cm d^{-1}.

Figure 4-13. Constructed wetland in Bainikeng in south China. The pond planted with *Nelumbo nucifera* Gearth is the third stage of a hybrid constructed wetland. Photo by Jan Vymazal.

Table 4-7. Treatment effect of a *Nelumbo* pond in a constructed wetland in Bainikeng, China. Results and data calculated from Wang et al. (1994) and Yang et al. (1994).

Parameter	**Inflow**	**Outflow**	**Removal**	**Inflow**	**Outflow**	**Removed**
	(mg l^{-1})		**(%)**	**(kg ha^{-1} d^{-1})**		
BOD_5	17.2	14.9	13	145	126	19
COD	60	56.2	6	506	474	32
SS	26.7	23.9	10	225	202	23
TN	21.9	20.2	8	185	170	18
TP	1.95	1.75	10	16.5	14.8	1.7
NH_4-N	14.9	14.7	1	126	124	2
NO_3-N	0.08	0.08	0	0.68	0.68	0

Constructed wetlands with floating-leaved macrophytes are very rare and there are no guidelines on how to design, operate and maintained these systems. Also, the results from existing systems are very limited and therefore, the usefulness of these systems is a questionable.

4.1.3 Systems with submerged macrophytes

Submerged aquatic macrophytes have their photosynthetic tissue entirely submerged (Fig. 4-13). Submerged plants, however, only grow well in oxygenated water and therefore cannot be used in wastewater with a high content of readily-biodegradable organic matter because the microbial decomposition of the organic matter will create anoxic conditions (Brix, 1993a). There is no direct evidence that short-term drops in oxygen concentration affect submerged plants distribution. However, longer exposure to low oxygen concentration may reduce growth of submerged macrophytes (Sahai and Sinha, 1976). Submerged macrophytes are often absent in anaerobic waters, but here also other factors like turbidity, sulfide and anaerobic sediments may play a major role (Best, 1982). In addition, the turbidity of the water must not be too high to prevent light transmission to the plants to support their photosynthetic activity (Reed et al., 1988). According to Westlake (1981) only changes in turbidity larger than the naturally occurring ones may cause detectable changes in growth, i.e., changes as large as 30%. However, Sculthorpe (1971) suggested that, for example, *Myriophyllum* is likely to be eliminated from areas where suspended solids tend to settle at high rates.

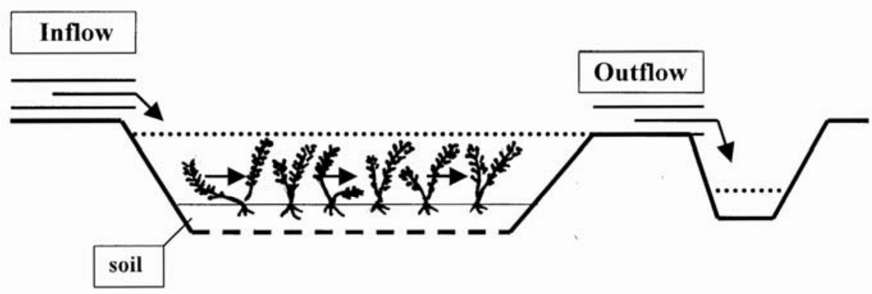


Figure 4-14. Schematic representation of a constructed wetland with submerged macrophytes. From Vymazal (2001a) with permission from Backhuys Publishers.

Various experiments have proven that minerals can be taken up directly by shoot tissues of submerged plants, however, there is also no question regarding the uptake capability of nutrients by the roots of these plants (e.g., Carignan and Kalff, 1980; Denny, 1980; Barko et al., 1991; Rattray et al., 1991). Most studies confirmed that although shoot uptake occurs, the major source for nutrients are sediments. Carignan and Kalff (1980) reported that even under hypertrophic conditions, the sediments contributed an average 72% of all the phosphorus taken up during growth. However, it has been shown that nutrient content of the submerged plants seem to be related to the concentrations in the water in which they grow (e.g., Best, 1977). Ozimek and Renman (1996) suggested that the extent uptake of nutrients from water and sediments varies widely among species. *Potamogeton pectinatus* (Sago pondweed), *Myriophyllum spicatum* (Eurasian watermilfoil), *Elodea canadansis* (Common waterweed) and *Ceratophyllum demersum* (Coontail) is the order in which uptake from water increases while the uptake from sediments decreases.

Although nutrient additions may initially stimulate growth of submerged macropyhtes (Goulder, 1969; Glanzer, 1974; Gnauck and Weise, 1976) ultimately macrophytic growth will become light-limited due to excessive growth of other primary producers which are more favorably placed with respect to the light source and which have higher relative growth rates, such as periphyton (Smith, 1969; Phillips et al., 1978; Sand-Jensen and Søndergaard, 1981; Sand-Jensen and Borum, 1983; 1991) and phytoplankton (Fitzgerald, 1969; Mulligan and Baranowski, 1969; Mulligan et al., 1976).

It has been demonstrated that epiphytic material absorbs PAR (photosynthetically active radiation) before reaching the leaf surfaces of vascular plants. The accumulation of epiphytic material may result in 80% attenuation of the incident radiation at the leaf surface (Twilley et al., 1985b). The development of epiphytic communities on the leaves of vascular plants may reduce net production through several mechanisms other than

PAR attenuation, including the reduction of diffusive transport of inorganic carbon, nitrogen and phosphorus. Reduction of photosynthesis of submerged plants was observed to correspond with increasing epiphytic material associated with nutrient enrichment.

Because of limiting conditions mentioned above (high nutrients, low light, high turbidity), the prime potential use of submerged macrophyte-based wastewater treatment systems is therefore for "polishing" secondary treated wastewaters, although good treatment of primary domestic effluent has been obtained in an experimental *Elodea*-based system in New Hampshire (Table 4-8, Eighmy and Bishop, 1988, Bishop and Eighmy, 1989). The authors reported that *E. nuttallii* (Nuttall waterweed) grew in primary effluent year-round, and effectively treated the wastewater. BOD_5 removals averaged 89.5% over the experimental period with effluent concentrations below 5 mg l^{-1}, while the controls without plants only removed an average of 66.8% of the influent BOD_5. The removal dropped off slightly during winter months, but was always > 78%. The authors concluded that the biofilm growing on the plant surfaces was primarily responsible for BOD_5 removal. The macrophyte systems were efficient at removing nitrogen and phosphorus from the wastewater (Table 4-8). During summer months nitrification was in excess of 90% in the macrophyte-based reactors, while in the control units it was generally less than 20%.

The treatment effect was comparable for *Elodea nuttallii* and *Myriophyllum heterophyllum* (Variableleaf watermilfoil) and superior to *Lemna minor* for BOD_5 and nitrogen. *E. nuttallii* grew well under all conditions tested, including water temperature ranging from 5°C to 25°C, over a wide range of photoperiods, and at influent wastewater strength of primary effluent. The macrophyte never experienced fouling by filamentous algae as other macrophytes did.

Table 4-8. Removal of BOD_5, nitrogen and phosphorus in experiments with various plant species. Seven different runs between December 1983 and December 1984. Data from Bishop and Eighmy (1989).

	BOD_5	TN	TP
Loading (kg ha^{-1} d^{-1})	2.50 – 63.7	0.54 – 9.2	0.09 – 2.18
Elodea nuttallii	77.6 – 97.4	17.3 – 70.9	1.60 – 44.5
Myriophyllum heterophyllum	89.5 – 95.9	19.7 – 70.8	18.9 – 38.1
Lemna minor	47.8 – 83.0	0.9 – 28.7	37.5 – 43.2
Plastic plants	65.5 – 92.8	0 – 34.1	20.2 – 23.0
Control (no plants)	25.9 – 83.6	0 – 15.0	0 – 42.0

Gu et al. (2001) reported good phosphorus removal from Everglades Agricultural Area runoff in Stormwater Treatment Area Cells (Fig. 4-15.) naturally colonized by submerged species *Ceratophyllum demersum* (Fig. 4-

15) and *Najas guadalupensis* (Southern naiad) (Fig. 4-16). The Outflow P concentrations were as low as 13-14 μg l^{-1} with removal effect of 61-70%.

Figure 4-15. Stormwater Treatment Area-1 Cell 4 (147 ha) in south Florida with *Ceratophyllum demersum* L. Photos by Jan Vymazal. STA-1 currently provides P removal from nutrient-rich water of the Everglades Agricultural Area before discharge into the Loxahatchee National Wildlife Refuge.

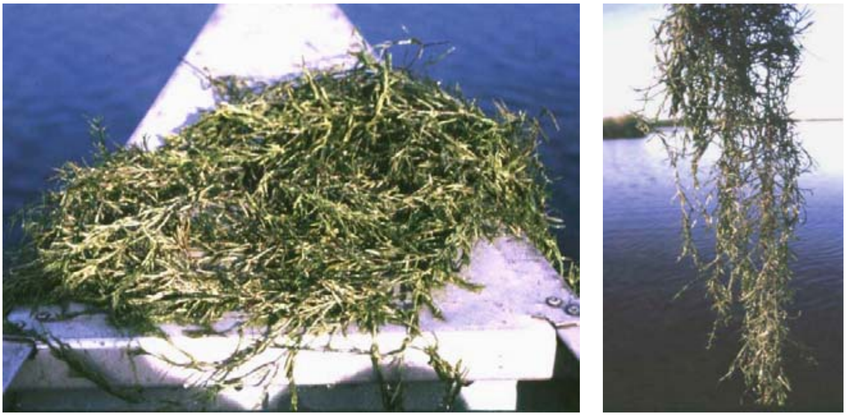


Figure 4-16. *Najas guadalupensis* (Spreng.) Magnus growing in the STA-1 in south Florida. Photos by Jan Vymazal.

Pietro (1998) reported that *Ceratophyllum demersum* also exhibits much higher phosphorus uptake rates as compared to other submerged species. For example, at P concentration of 0.2 mg L^{-1}, Pietro (1998) found the uptake rate of 284 mg m^{-2} d^{-1} while DeBusk et al. (1989) found only 27 mg m^{-2} d^{-1} in *Egeria* spp. *C. demersum* is a plant without true root system and the nutrients are taken up through the stem and leaves of the plant. This may be the explanation why *C. demersum* takes phosphorus more efficiently than other species (Pietro 1998). However, it has been reported that *Najas guadalupensis* (Southern waternymph) stored nearly 6 times more (1.42 g P m^{-2}) P than *C. demersum* and periphyton during an outdoor tank experiments in Florida (SAV, 1999).

Reddy et al. (1982) reported the use of *Egeria densa* (Brazilian elodea) in a series of ponds designed for the treatment of agricultural drainage waters. Uptake rate of *Eggeria* in the second pond in the series was estimated at 3.5 g N m^{-2} yr^{-1} and 0.5 g P m^{-2} yr^{-1} The uptake rate was much lower that that of *Eichhornia* but comparable with *Typha latifolia* (Common cattail). The authors pointed out that submerged plants, similarly to free-floating species, must be harvested periodically to maintain nutrient removal efficiencies.

Hammer and Knight (1992) reported the use of constructed wetland with submerged vegetation as a part of a constructed wetland that polishes effluents from primary and secondary lagoons for a 500-hog production facility at Pontotoc, Mississippi. Cell 1 has *Typha latifolia* in the shallow regions and *Potamogeton pectinatus* (Sago pondweed) in the deeper regions; cell 2 has *Eleocharis dulcis* (Spikerush) in the shallow regions. The inflow loadings were 533 and 964 g NH_3-N m^{-2} yr^{-1} for cells 1 and 2, respectively. The respective removed amount were 397 and 539 g NH_3-N m^{-2} yr^{-1}, resulted in 74.5 and 55.9% removal and effluent NH_3-N concentrations of 27.5 and 48 mg L^{-1}.

Hammer and Knight (1992) also reported the use of submerged macrophytes – *Potamogeton pectinatus* and *Vallisneria americana* (Wild celery) within a large treatment wetland at Minot, North Dakota. The Minot wetlands system provides polishing for wastewater from about 35,000 residents. The design includes four cells 177 m wide and 745 m long with a marsh(A)-pond(B)-marsh(C)-pond(D)-marsh(E) zones located sequentially within each cell. Marsh zones A and E are planted with *Typha latifolia* with an operating depth of 15 cm, marsh zone C is planted with *Scirpus validus* and *Lemna* with operating depth of 30 cm and pond zones B and D are planted with submerged species *Potamogeton pectinatus* and *Vallisneria americana* (Eel-grass) with an operating depth of 60 cm. The treatment performance of the system was very good: BOD_5: inflow 18.8 mg l^{-1} - outflow 3.4 mg l^{-1}; TSS: 50.3 mg l^{-1} - 8.6 mg l^{-1}; ammonia: 2.1 mg l^{-1} - 0.2 mg l^{-1}.

Vincent (1992) reported the use of a treatment wetland system which purifies water from a swimming area of the Lac des Régates in Montréal, Canada. The wetland systems consist of four ponds with submerged species present in three of then. The first pond is planted with *Eichhornia crassipes* and *Hydrocharis morsus-ranae* (Frog's bit), the second pond is planted with emergent species *Iris versicolor* (Blue flag), *Phragmites australis*, *Scirpus acutus* (Hardstem bulrush) and *Typha latifolia*. The third pond is planted with emergent species *Pontederia cordata* (Pickerelweed) and submerged species *Myriophyllum spicatum* (European watermilfoil), *Elodea canadensis* and *Alisma triviale* (Northern water plantain). The last pond was planted with submerged species only: *Elodea canadensis*, *Potamogeton pectinatus* and *Vallisneria americana*. The treatment system was quite efficient in removing orthophosphate and nitrate. However, removal of total P, ammonia and suspended solids was quite low. Also, there was no removal of fecal coliforms (Table 4-9). However, the inflow concentrations of these parameters were quite low.

Table 4-9. Treatment performance of the Lac des Régales wetland system during the period 1990-1991. Data from Vincent (1992).

Parameter	**Unit**	**Inflow**	**Outflow**
PO_4^{3-}	μg l^{-1}	1.31	0.35
TP	μg l^{-1}	14.4	14.0
NO_3^-	μg l^{-1}	57	10.6
NH_4^+	μg l^{-1}	6.3	5.2
SS	mg l^{-1}	1.85	1.80
FC	CFU 100 ml^{-1}	41	74

Bavor et al. (1987, 1989) compared various types of constructed wetlands treating domestic sewage from a conventional trickling filter/maturation pond in Hawkesbury, Australia. Systems consisted of lined trenches planted with *Typha orientalis* (Cattail) or *Scirpus validus* (Soft-stem bulrush) in gravel and submerged macrophyte *Myriophyllum aquaticum* (Parrot's feather) growing in open water with no solid matrix. The treatment effect was quite good, however, the *Myriophyllum* system was found to be the least effective design format tested.

Toet et al. (2005) reported on the use of submerged macrophytes in combination with emergent species in a constructed wetland at Everstekoog on the island of Texel, The Netherlands. The wetland system had a total water surface of 1.3 ha divided into 9 parallel ditches. The first half of eight ditches was 0.2 m deep and contained *Phragmites australis* or *Typha latifolia*, while the second half was 0.4 m deep and contained submerged macrophytes, mainly *Elodea nuttallii*, *Ceratophyllum demersum* and *Potamogeton* spp. (Fig. 4-17). The system achieved 26% reduction of nitrogen (126 g N m^{-2} yr^{-1}) and less than 5% of phosphorus (5 g P m^{-2} yr^{-1}). Nitrogen removal was highest in the shallow front sections planted with macrophytes, largely owing to the denitrification. The increase of in the oxygen concentrations predominantly occurred in the rear, deeper sections due to the presence of submerged macrophytes and associated periphytic algae.

Figure 4-17. Constructed wetland Everstekoog on the island of Texel, The Netherlands. From the right, ditches 1, 3, 5 and 7 are planted with *Phragmites*, ditches 2, 4, 6 and 8 are planted with *Typha* (at the time after a harvest). The second half of the ditches contains submerged macrophytes. Photos by Ruud Kampf, with permission.

The use of submerged macrophytes for the wastewater treatment is still in the experimental stage, with species like *Egeria densa*, *Elodea canadensis*, *Elodea nuttallii*, *Ceratophyllum demersum, Hydrilla verticillata* (Hydrilla), *Cabomba caroliniana* (Fanwort), *Myriophyllum heterophyllum* and *Potamogeton* spp. (Pondweeds) being the most promising (McNabb, 1976; Reed et al., 1988; Lakatos, 1998). This category may also include systems with macrophytic algae from the order Characeae, *i.e*., *Chara* and *Nitella* (Smith and Kalin, 1988, 1989).

4.1.4 Systems with emergent macrophytes

A typical free water surface constructed wetland (FWS CW) with emergent macrophytes (Fig. 4-18) is a shallow sealed basin or sequence of basins, containing 20-30 cm of rooting soil, with a water depth of 20-40 cm. Dense emergent vegetation covers significant fraction of the surface, usually more than 50%. Besides planted macrophytes naturally occurring species may be present (Kadlec, 1994). The most commonly used species for FWS constructed wetlands are in Europe: *Phragmites australis* (Common reed), *Scirpus lacustris*; North America: *Typha* spp. (Cattail), *Scirpus* spp. (Bulrush), *Sagittaria latifolia* (Arrowhead); Australia and New Zealand: *Phragmites australis*, *Typha* spp., *Bolboschoenus* (*Scirpus*) *fluviatilis* (Marsh clubrush), *Eleocharis sphacelata* (Tall spikerush), *Scirpus tubernaemontani* (= *Scirpus validus*, Soft-stem bulrush).

Flow is directed into a cell along a line comprising the inlet (Fig. 4-19), upstream embankment, and is intended to proceed to all parts of the wetland to one or more outlet structures (Figs. 4-20 and 4-21). The shallow water depth, low flow velocity, and presence of the plant stalks and litter regulate water flow and, especially in long, narrow channels, ensure plug-flow conditions (Reed et al., 1988). One of their primary design purposes is to contact wastewater with reactive biological surfaces (Kadlec and Knight, 1996).

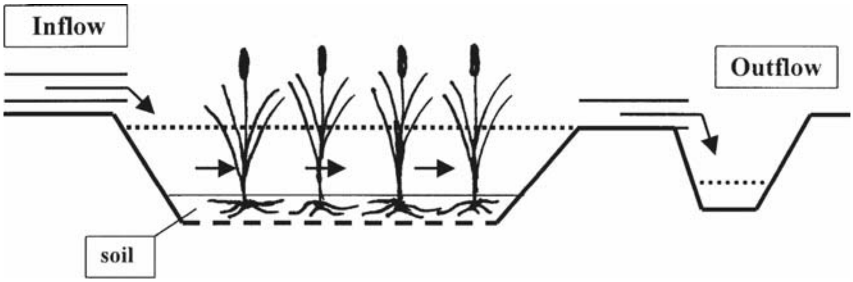


Figure 4-18. Schematic representation of the free water surface constructed wetland with emergent macrophytes. From Vymazal (2001a) with permission from Backhuys Publishers.

Figure 4-19. Examples of water distribution at the inflow of FWS constructed wetlands. Top: Casino, NSW, Australia; bottom: Pensacola, Florida, USA. Photos by Jan Vymazal.

Figure 4-20. Outflow structure in a stormwater treatment constructed wetland, Brisbane, NSW, Australia. Photo by Jan Vymazal.

Figure 4-21. The H-flume structure for the outflow control from Lakeland FWS constructed wetland in Florida. From U.S. EPA (1993), with permission.

Free water surface constructed wetlands with emergent macrophytes function as land-intensive biological treatment systems. Inflow water containing particulate and dissolved pollutants slows and spreads through a large area of shallow water and emergent vegetation (Kadlec and Knight, 1996). Suspended solids removal is usually a fairly rapid physical process. The major removal mechanisms are sedimentation, aggregation and surface adhesion (QDNR, 2000). The largest and heaviest particles will predominantly settle out in the inlet open water zone while slightly smaller and lighter particles may only settle out after flowing into wetland vegetation. Wetland vegetation promotes this enhanced sedimentation by reducing water column mixing and resuspension of particles from the sediment surface. Sedimentation and resuspension are opposing processes with the balance controlled by water column mixing and turbulence at the sediment-water interface (QDNR, 2000).

Aggregation is a process whereby particles naturally tend to flocculate. The degree to which aggregation will occur is determined by a balance between particle attraction (controlled by surface chemistry characteristics) and the strength of the shear forces on the particles. Shear forces within the water column are related to mixing and turbulence (QDNR, 2000).

The smallest of particles such as clay colloids, may not aggregate enough to settle out in the detention time available in a wetland. For these particles the only removal mechanism available is adhesion onto surfaces within the water column. The main surfaces within the water column are the emergent macrophytes and the biofilms growing on the surface of these plants (QDNR, 2000).

Settleable organics are rapidly removed in FWS systems under quiescent conditions by deposition and filtration. Attached and suspended microbial growth is responsible for the removal of soluble organic compounds which are degraded aerobically as well as anaerobically (see section 2.2). The decomposition pathway by which wetland carbon loads are processed is determined by a balance between the carbon load and the supply of oxygen. Oxygen is supplied to the wetland water column by diffusion through the air-water interface and via the photosynthetic activity of plants in the water column, namely periphyton and benthic algae (Kadlec et al., 2000; QDNR, 2000).

FWS treatment wetlands typically have aerated zones, especially near the water surface because of atmospheric diffusion, and anoxic and anaerobic zones in and near the sediments. In heavily loaded FWS wetlands, the anoxic zone can move quite close to the water surface. Biomass decay provides a carbon source for denitrification, but the same decay competes with nitrification for oxygen supply. Low winter temperatures enhance oxygen solubility in water, but slow microbial activity (Kadlec and Knight, 1996).

Nitrogen is most effectively removed in FWS constructed wetlands by nitrification/denitrification. Ammonia is oxidized by nitrifying bacteria in aerobic zones, and nitrate is converted to free nitrogen or nitrous oxide in the anoxic zones by denitrifying bacteria. Volatilization is likely as both plankton and periphyton algae grow in FWS CWs and higher pH values during the day may be favorable for ammonia loss. Billore et al. (1994) reported that ammonia loss through volatilization from FWS wetland system in India averaged 101 mg m^{-2} d^{-1}. The ammonia loss was lower as compared to Duckweed system (125 mg m^{-2} d^{-1}) but higher than system without plants (41 mg m^{-2} d^{-1}).

FWS CWs provide sustainable removal of phosphorus, but at relatively slow rates. Phosphorus removal in FWS systems occurs from adsorption, absorption, complexation and precipitation. However, precipitation with Al, Fe and Ca ions - is limited by little contact between water column and the soil (Kadlec and Knight, 1996; Vymazal et al., 1998a). Substantial amounts of nitrogen and phosphorus may be stored in the peat/litter compartment. Algal and microbial uptake may be high but this retention is a short-term process and nutrients are washed out from the detritus back to the water.

Macrophyte uptake as removal mechanisms in FWS CWs is restricted by the fact that vegetation is not regularly harvested. Crites (1992) recommended vegetation harvest in FWS constructed wetlands only every 3 to 5 years. Results from a FWS constructed wetland at Listowel in Ontario indicate that emergent vegetation is not the major pool for nitrogen removal (Herskowitz, 1986). According to this study, only 10.2% and 5.62% of the total removed nitrogen and phosphorus, respectively, was removed by harvesting of *Typha* during years 1981-1984. Hurry and Bellinger (1990) reported that the harvesting of aerial biomass of *Phalaris arundinacea* in FWS wetland in England would have removed 49.4 g N m^{-2} yr^{-1} and 10.9 g P m^{-2} yr^{-1}, representing 11% removal of the annual nitrogen loading and 7% removal of the annual phosphorus loading. Hunt et al. (2002) found 8.1% and 8.4% of the inflowing nitrogen from swine wastewater sequestered in the aboveground biomass of *Typha* spp. and *Scirpus* spp., respectively. Belowground litter formation is important since represents an amount of nutrients that have been transferred from a rapidly circulated inorganic form to organic compounds with a slower circulation rate. Some of these nutrients will be subject to long-term accumulation in the soil, hence contributing to the wastewater-treatment capacity of the system (Sundblad and Wittgren, 1989).

Wetlands are known to offer a suitable combination of physical, chemical and biological factors for the removal of pathogenic organisms (Table 4-10). Physical factors include mechanical filtration, exposure to ultraviolet radiation, and sedimentation. Chemical factors include oxidation, exposure to biocides excreted by some wetland plants, and absorption by organic matter. Biological removal mechanisms include antibiosis, predation by nematodes and protists, attack by lytic bacteria and viruses, and natural die-off (Gersberg et al., 1989a).

Zdragas et al. (2002) pointed out that solar radiation plays a very important role in reduction of coliform populations. The mid-UV (UV-B, 290-320 nm) and the near-UV (UV-A; 320-400 nm) have the most detrimental effects on coliforms, although harmful effects can be caused by wavelengths up to 700 nm (Nasim and James, 1978; Jagger, 1983). These wavelengths have both lethal and non-lethal effects on bacteria and the lethal effects are primarily caused by lesions in the DNA. The harmful effect of UV light centers on the formation of thymine dimeres on strands of DNA, although other types of damage may occur. Non-lethal effects include growth delay and inhibition, inhibition of induced enzyme synthesis, reduced active transport, and mutagenesis (Maki, 1993). Additionally, exposure to solar radiation can lead not only to bacteria but also to virus inactivation or removal (Bitton, 1980).

Table 4-10. Removal mechanisms for microbial pollution in FWS wetlands with emergent macrophytes. Adopted from QDNR (2000), with permission.

Mechanism	**Description**
Sedimentation	Trapping pathogens in the sediments.
Surface adhesion	Trapping pathogens on biofilms primarily supported by emergent aquatic macrophytes.
Aggregation	Natural processes resulting in the clumping of particles. Aggregation is enhanced in wetlands because wetland vegetation reduces water column mixing forces that act to break up aggregation.
Natural die-off and competition	Exposure to hostile environmental conditions, particularly high and variable temperature, dissolved oxygen, redox and pH. For pathogens trapped in biofilms in particular, competitions for resources with the natural microbiota is a factor in die-off. Part of this competition may be the excretion of substances that are toxic to pathogens and other organisms (allelopathy).
Predation	Many pathogens may be eaten by filter feeding zooplankton. Even where ingestion does not kill the pathogen, it effectively packages the pathogens into fecal pellets that are more easily removed from the water column.
UV exposure	Exposure to strong sun light and UV radiation can damage proteins and genetic material in many organisms. Enteric bacteria and viruses are the most susceptible to this mechanism.

In Germany, during the 1960s, Dr. Käthe Seidel tried to improve the performance of rural and decentralized wastewater treatment which was either septic tanks or pond systems with inefficient cleaning results. She planted helophytes into the shallow embankment of tray-like ditches and created artificial trays and ditches grown with helophytes. The system worked and results were reported. Seidel named this early system the hydrobotanical system. However, the concept to apply higher plants to sewage treatment was already difficult to understand for sewage engineers and technicians who had eradicted any visible plants on a treatment site for more than 50 years (Börner et al., 1998).

In spite of many prejudices among civil engineers about odour nuisance, attraction of flies, poor performance in cold periods the IJssel Lake Polder Authority in Flevoland in The Netherlands constructed its first FWS constructed wetland in 1967 (Veenstra, 1998). The wetland had a design depth of 0.4m and the total area was 1 hectare. A star shape layout was chosen in order to obtain optimum utilization of the available area which, however, complicated macrophyte harvesting and maintenance in general (de Jong, 1976). Therefore, longitudinal channels were added (Fig. 4-22) to facilitate mechanical biomass harvesting and system maintenance. The new wetland design included channels of 3m wide and 200m long, separated by parallel stretches of 3m resulting in an increase in land requirement from $5m^2$ per PE for the star arrangement to $10m^2$ per PE. The system exhibited a very good treatment effect and in early 1970s, about 20 FWS of this ditch type, called planted sewage farm (or Lelystad process) were in operation in The Netherlands (Greiner and de Jong, 1984; Veenstra, 1998).

Figure 4-22. FWS constructed wetland at Lelystad, the Netherlands, planted with *Scirpus lacustris*. Photo by Hans Brix, with permission.

In 1968, a FWS CW was created in Hungary near Keszthely in order to preserve the water quality of Lake Balaton and to treat wastewater of the town (Lakatos, 1998). The constructed wetland was established in place of an existing natural wetland in peat soil. The existing natural wetland consisted of six ponds 40-60 cm deep with a surface area of 10 ha and the ponds were fed with 8000 m^3 d^{-1} of mechanically pre-treated wastewater. Contrary to the North America situation, FWS CWs did not spread throughout Europe significantly while CWs with sub-surface flow (HF systems) drew much more attention at the end of the 20th century (Vymazal et al., 1998a). However, FWS CWs are in operation in many European countries, e.g. Sweden, Denmark, Poland, Ireland, Estonia or Belgium. In Sweden, FWS systems were constructed with nitrogen removal as a primary target but other aims, e.g. biodiversity, irrigation are also taken into consideration (Vymazal et al., 2006a). Sometimes, the aim is to provide phosphorus polishing after chemical treatment and a buffer in case of treatment failure in the conventional treatment plant (Sundblad, 1998). More than 2,350 ha of wetlands have been created in Sweden in the agricultural landscape between 1996 and 2002 with different kinds of subsidies and in Denmark about 3,200 ha have been created until 2004 (Vymazal et al., 2006a).

In North America, the FWS technology started with the ecological engineering of natural wetlands for wastewater treatment. Between 1967 and 1972, Howard T. Odum of the University of North Carolina, Chapel Hill, began a study using coastal lagoons for recycling and reuse of municipal wastewaters (Odum, 1985). In 1972, Odum, who had relocated to the University of Florida in Gainesville, began with Katherine Ewel to study the effectivness of natural cypress wetlands for municipal wastewater recycling (Odum et al., 1977; Ewel and Odum, 1984). At about the same time, R.H. Kadlec and co-workers at the University of Michigan in Ann Arbor began the Houghton Lake project, the first in-depth study using engineered wetlands for wastewater treatment in a cold climate region (Kadlec et al., 1975, Kadlec and Tilton, 1979). Industrial stormwaters and process waters were also applied to constructed wetland/pond system in 1975 at Amoco Oil Company's Mandan Refinery in North Dakota (Litchfield and Schatz, 1989). The use of FWS constructed wetlands for urban stormwater treatment was pioneered in California in the early 1980s (Chan et al., 1982; Meiorin, 1989) and mine wastewater treatment wetlands have also been used since the early 1980s (Kleinmann, 1985; Brodie et al., 1988; Wieder, 1989). At present, there are hundreds, and probably thousands of FWS constructed wetlands with emergent macrophytes treating municipal and industrial wastewaters, agricultural wastes and runoff, mine drainage waters and stormwaters in North America (Pries, 1994; Kadlec and Knight, 1996; Kadlec, 2003).

Free water surface CWs are also commonly used in Australia and New Zealand, especially for treatment of municipal wastewater, stormwater and pasture runoff (Greenway and Simpson, 1996; Greenway and Wooley, 1999; QDNR, 2000; Tanner et al., 2000; 2005a,b; Geary et al., 2006).

4.1.4.1 Sizing

In the literature, various relationships have been developed in order to assess the wetland area necessary to produce required water quality. Each regulated parameter gives rise to a wetland area necessary for the reduction of that pollutant to the required level. The required wetland area is the largest of the individual required areas. It has been shown that the order of size necessary for effective removal of various pollutant is in the order SS < BOD < nitrogen < phosphorus (Kadlec and Knight; 1996).

Design methods can be based on either volume or area (Wallace and Knight, 2006). Volume-based methods use a hydraulic retention time to assess pollutant reduction (Reed et al., 1995; Crites and Tchobanoglous, 1998), area-based methods assess pollutant reduction using the overall wetland area (Kadlec and Knight, 1996; Economopoulou and Tsihrintzis, 2004). Kadlec and Knight (1996) suggested a plug-flow k-C* model:

$$\ln [(C_o - C^*)/C_i - C^*)] = -k_A/q \qquad (4.1)$$

where C_o = outlet concentration (mg l^{-1})
C_i = inlet concentration (mg l^{-1})
C^* = background concentration (mg l^{-1}) (*see footnote*)
k_A = first order areal rate constant (m d^{-1})
q = hydraulic loading rate (m d^{-1})

Hydraulic loading q could also be expressed as:

$$q = Q/A \qquad (4.2)$$

where Q = average wastewater flow (m^3 d^{-1})
A = wetland area (m^2)

Combining Equations 4.1 and 4.2, the wetland area could be calculated as:

$$A = Q \ln [(C_i - C^*)/C_o - C^*)] / k_A \qquad (4.3)$$

Wetland ecosystems typically include diverse autotrophic and heterotrophic components. Most wetlands are more autotrophic than heterotrophic, resulting in a net surplus of fixed carbonaceous material that is buried as peat or is exported downstream to the next system (Mitsch and Gosselink, 1993). This net production results in an internal release of particulate and dissolved biomass to the wetland water column, which is measured as non-zero of BOD, SS, TN and TP. These wetland background concentrations are denoted by C*. Treatment wetlands background concentrations ranges can be estimated from systems that are loaded at a low enough rate to result in asymptotic concentrations along gradient of increasing distance from the inflow. Kadlec and Knight (1996) estimated following background concentrations for FWS wetlands (in mg l^{-1} except bacteria): BOD_5: 3.5 + $0.053C_i$; TSS: 5.1 + $0.16C_i$; organic N: 1.50; NH_4-N: 0.0; NOx-N: 0.0 (if sequential with ammonia-N), TN: 1.50; TP: 0.02; fecal coliforms: 300 (CFU 100 ml^{-1}).

The constant k_A is temperature dependent (Kadlec, 1994):

$$k_A = k_{20}\,\Theta^{(T-20)} \tag{4.4}$$

where k_{20} = first order areal rate constant at 20°C (m d^{-1})
Θ = temperature coefficient for rate constant

Kadlec and Knight (1996), based on the large literature dataset, estimated following values of k_A (m d^{-1}): BOD_5: 0.093 (Θ = 1.00), TSS: 2.74 (Θ = 1.00), organic N: 0.047 (Θ = 1.05), NH_4-N: 0.049 (Θ = 1.04), NO_x-N: 0.096 (Θ = 1.09), TN: 0.06 (Θ = 1.05), TP: 0.033 (Θ= 1.00), Fecal coliforms: 0.21(Θ = 1.00).

Regression equations may also be used to describe the removal performance; these form a second method of setting wetland size for some variables. These regressions have unsatisfyingly low correlation coefficients because they span a large number of unquantified system variables (Kadlec and Knight, 1996). However, Wallace and Knight (2006) reported the use of polynomic inflow load vs. outflow concentration curve fits based on an assumed first-order background concentration (k-C*) model. The loading charts were based on a large world-wide dataset that define 50%. 75% and 90% of the data points for that data set. For example, FWS wetland loaded at 30 kg BOD ha^{-1} d^{-1} should meet 25 mg l^{-1} at the outflow 90% of the time while 25 mg l^{-1} at the outflow will be met 50% of the time when loaded at 68 kg ha^{-1} d^{-1}. The charts were prepared for BOD_5, TSS, TKN, TP and fecal coliforms. To meet outflow concentrations of 30 mg l^{-1} of BOD_5 and TSS 90% of the time, FWS wetlands should be loaded with up to 60 kg ha^{-1} d^{-1} and 70 kg ha^{-1} d^{-1}, respectively. To meet 10 mg l^{-1} TKN at the outflow 90% of the time, the loading should not exceed 15 kg TKN ha^{-1} d^{-1}.

4.1.4.2 Municipal wastewater

FWS constructed wetlands with emergent macrophytes (Figs. 4-23, 4-24) are very commonly used to treat municipal wastewater, especially as a tertiary treatment stage (Table 4-11). Kadlec (2003) pointed out that the value of tertiary wetlands includes ancillary benefits, such as wildlife habitat and public use (U.S. EPA, 1993; Knight et al., 2001).

Removal efficiency in terms of percentage may not be high all the time due to low inflow concentrations in many systems but FWS constructed wetlands provide effluents with low concentrations of organics and suspended solids. Removal of nitrogen and phosphorus is highly variable but usually amounts to about 50%. Removal of fecal coliforms varies between one ond two orders of magnitude (Table 4-11). For detailed information see Kadlec and Knight (1996), Kadlec et al. (2000).

Figure 4-23. FWS constructed wetland for tertiary treatment of municipal sewage in Pöltsamaa, Estonia planted with *Typha latifolia*. Photo by Jan Vymazal.

Figure 4-24. A 3-ha FWS constructed wetland for tertiary treatment of municipal sewage in Casino, NSW, Australia, planted with *Bolboschoenus fluviatilis* (picture) and *Typha* spp. Photo by Jan Vymazal.

Table 4-11. Treatment performance of FWS constructed wetlands with emergent vegetation for treatment of municipal sewage. Chemical parameters in mg l^{-1}, fecal coliforms (FC) in log CFU 100 ml^{-1}. Values are mean values for several years of performance.

Location	**Country**	**Area**	**Flow**	**BOD_5**		**TSS**		**TP**		**TN**		**NH_4-N**		**FC**		**Ref.**
		(m^2)	**(m^3 d^{-1})**	**In**	**Out**	**In**	**Out**	**In**	**Out**	**In**	**Out**	**In**	**Out**	**In**	**Out**	
Warangal	IND	118	5.0	152	19.5	165	16	7.4	1.7	36	3.9	20	3.6			1
Listowel 2	CAN	909	26	19.6	11.3	23	9.0	1.05	0.76	12.2	8.1	7.2	5.1	4.3	4.2	2
Kohukoku	NZ	1,200	30	48	7	107	16	14.9	10.4	58	27.3	46.9	19.9	4.4	2.9	3
Portland	NZ	1,300	68	32	10	111	15	3.5	3.2	10.4	5.9	0.7	3.3	3.4	3.0	3
Listowel 4	CAN	1,324	27	56	9.6	111	8.0	3.2	0.6	19.1	8.9	8.6	7.9	5.3	3.0	2
Cairns	AUS	1,683	84.2	9.0	4.0	5.0	4.0	7.8	6.9	6.1	1.5	0.3	0.2	4.9	3.0	4-6
Deurle	BEL	3,060	543	46	10.9	34	7.7	3.1	2.3	24.7	17.2					7
Village of Alfred	CAN	3,120	57.5	3.6	2.5	83	17.5	0.33	0.04	1.7	0.9	0.11	0.06	1.9	1.5	8
Pompia, Crete**	GRE	5,500	144	165	7.8	208	6.2	14.8	7.45	34.2	17.9			4.4	2.8	9
Ingham	AUS	7,920	317	22	11	24	16	6.8	5.4	19.5	9.7	8.0	5.4			4-6
Beijing	CHN	10,638	500	125	17.8	275	17	0.94	0.42	14.4	5.1					10
Põltsamaa*	EST	12,100	1,050	46	13			5.5	4.1	21	12	9.7	6.7			11
Wiezyca	POL	18,370	129	225	9.9	194	10	4.6	0.65	31.6	4.4	8.2	1.5			12
River Herbert	CAN	19,000	169	21	5.0	22	12	3.2	0.55	19.9	4.5			4.3	2.6	13
Frombork	POL	22,000	850	107	45	197	65	11.5	11.0	49	37	38.7	27			12
Arlington	USA	34,400	1,457	14.4	10.8	85	81	0.85	0.63			0.13	0.08			14
South Lismore	AUS	36 000	3,500-20,000	9.4	0.7	74	1.8	2.8	0.2	5.4	1.0	0.7	0.02	3.1	2.7	15
Cambridge	NZL	66,000	3,234	52	13	40	13	12.6	12.1			47	25			16
Lake Coral	USA	210,000	1,477	11.6	2.6	6.0	1.5	6.2	5.2	20	1.6	7.7	0.3			17
Oxelösund*	SWE	230,000	5,000	17	3.6			0.4	0.04	23	15					18

*BOD_7 (=cca. 1.15 BOD_5), 1-Jayakumar and Dandigi (2002), 2-Herskowitz (1986), 3-Tanner et al. (2000), 4-Greenway and Wooley (1999), 5-QNDR (2000), 6-Vymazal et al. (2006a), 7-VMM (2006), 8-Cameron et al. (2003), 9-Dialynas et al. (2002)**TKN values, 10-Xianfa and Chuncai (1994), 11-Mander et al. (2001), 12-Kowalik and Obarska-Pempkowiak (1998); Obarska et al. (1994), 13-Hanson (2002), 14-Kadlec and Knight (1996), 15-Davison et al. (2006), 16- T. Headly and C. Tanner (pers. comm.), 17-Knight et al. (1985), 18-Sundblad (1998), Anderson et al. (2005)

4.1.4.3 Agricultural wastewaters

The use of constructed wetlands for treating concentrated animal wastes is a relatively new idea (Figs. 4-25, 4-26). Knight et al. (2000) pointed out that the majority of the wetlands engineered for livestock wastewater treatment are small, with an average system size of 0.6 ha. The majority of the swine, poultry and dairy treatment wetland systems are less than 0.1 ha. All of the wetland systems have some form of pretreatment (Fig. 4-25). The most common form of pretreatment is a settling basin or anaerobic lagoon. Examples of removal efficiency of constructed wetlands for animal waste treatment are presented in Table 4-12, detailed information and reviews can be found in DuBowy and Reaves (1994), Kadlec and Knight (1996), Cronk (1996), Payne and Knight (1997), Hunt et al. (1995, 1997, 2002), Knight et al. (2000), Hunt and Poach (2001), Stone et al. (2002).

Wetlands for animal wastewater treatment should always be coupled with additional waste management strategies. The accumulation of solids shortens the effective life of a constructed wetland, making solids removal a necessary pretreatment step (Cronk, 1996). Also, animal wastewaters are highly concentrated and, therefore, some degree of dilution prior to the discharge to a constructed wetland may be necessary.

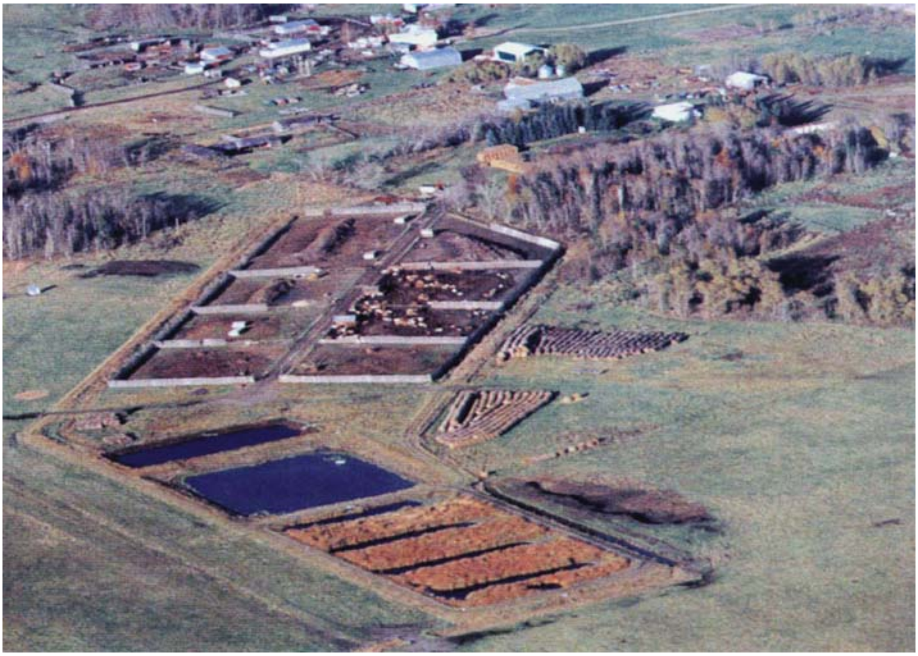


Figure. 4-25. A FWS livestock constructed wetland for high strength runoff at the 300-head cow-calf feedlot in Riverton, Manitoba, Canada. The system consists of three units: the settling pond, the holding pond, and 0.5 ha treatment wetland. Photo provided by Prairie Farm Rehabilitation Administration, Winnipeg, BC, Canada.

Figure 4-26. Integrated constructed wetland for farm yards wastes treatment in Anne Valley, Ireland. Top – general view, bottom – detail. Photos by Lenka Kröpfelová.

Table 4-12. Examples of FWS constructed wetlands with emergent vegetation for treatment of agricultural wastewaters.

Waste	Ref.	Location	Size (m^2)	Flow ($m^3 d^{-1}$)	BOD_5 ($mg l^{-1}$)		COD ($mg l^{-1}$)		TSS ($mg l^{-1}$)		TP ($mg l^{-1}$)		TKN ($mg l^{-1}$)		NH_4-N ($mg l^{-1}$)		FC (log CFU 100 ml^{-1})	
					In	Out	In	Out	In	Out	In	Out	In	Out	In	Out	In	Out
Swine	1	AL, USA	4,056	26.1	77	7.9	320	64	136	15.5	28.4	6.8	74	12.2	56	8.6	5.1	3.8
Swine	2	AL, USA	5,700	31.3	64	6.1			105	9	25.8	6.2	70	6	55	3.5	5.2	3.0
Swine[a]	3	NC, USA	440	9.94			808	464	363	235	73	55	175	109				
Dairy	4	IN, USA	1,850	0.75	910	68			483	31	25.3	4.2	215	30.4	199	21.6	2.2	0.8
Dairy	5	MS, USA	432	1.36	31	7.2	294	101	120	43	16.8	7.4			6.8	1.7	4.2	2.9
Dairy	6	OR, USA	882	34.8	705	242	1,628	655	542	142	33	17			126	65	6.1	5.2
Dairy	7	MD, USA	1,700		1,914	59			1,645	65	52.6	2.1			72	32		
Dairy	8	CT, USA	416	2.69	2,683	611			1,284	130	25.7	14.1	103	74	7.7	52.3	5.8	4.1
Dairy	9	ONT,CAN	600		357	202			1,596	48	25	3.9	119	17.5	50	12	6.0	4.1
Poultry	10	AL, USA	781	2.5	198	101	350	180			26	17.1	117	68	95	62		
ICW[b]	11	Ireland	8,643		1,075	18.6	2,126	49	660	21	23.3	0.5			78	0.65	5.1	2.4

1-McCaskey and Hannah (1997), 2-Hammer et al. (1993), 3-Poach et al. (2004) [a]values given for TN, not TKN, 4-Reaves and DuBowy (1997), 5-Cooper and Testa (1997), 6-Moore and Niswander (1997), 7-Schaafsma et al. (2000), 8-Majer Newman et al. (2000), 9-Hermans and Pries (1997) 10-Hill and Rogers (1997), 11-Carroll et al. (2005) [b]data for PO_4-P and *E.coli*, average values for 12 systems; ICW=integrated constructed wetlands treat yard and dairy washings, rainfall along with silage and manure effluents (see Fig. 4-26).

4.1.4.4 Stormwater runoff

Constructed wetlands for stormwater runoff treatment improve water quality, modify flow rates by storing water temporarily in shallow pools (especially important in arid and semi-arid areas) that create growing conditions suitable for emergent vegetation, attenuate flow, and reduce downstream scouring and erosion (Fig. 4-27).

Stormwater runoff from agricultural fields (Figs. 4-28, 4-29), particularly after spreading fertilizer, results in high nutrient loadings to the receiving streams. Routing of stormwater runoff and tile drainage to a constructed wetland can provide treatment prior to discharge to the environment and the potential to recycle the nutrients by spray irrigating the collected water back onto the fields where it originated (Pries, 1994; Braskerud, 2002).

Target pollutants in urban and road stormwater runoff constructed wetlands are suspended solids, organics, oil and grease and heavy metals. Target pollutants in systems designed to treat agricultural runoff, pasture runoff or golfcourses are primarily nutrients (Table 4-13).

Pontier et al. (2004) pointed out that road runoff is a highly variable and intermittent feedstock, which contrasts to more uniform or predictable flows and loadings of the other wastewaters treated using constructed wetlands. There have been several promising studies on the use of wetlands for road runoff (e.g., Nix et al., 1988; Mungur et al., 1995; Lee et al., 1997a, b; Shutes et al., 1997; Osterkamp et al., 1999; Carpeto and Purchase, 2000), but as yet no established design criteria have emerged.

The specific design concept has not been developed mainly because of very variable flow and quality of the runoff. Most of the early design recommendations provided “rule-of-thumb” approaches to preliminary sizing of wet detention basins to maximize the removal of pollutants.

The Ontario Ministry of Environment and Metropolitan Toronto and Region Conservation Authority (1992) recommended that stormwater wetlands are be determined as five percent of the catchment area. Schueler (1992) indicated that a smaller wetland area of one percent of the total watershed area was considered acceptable for wetlands with deep zones and longer hydraulic residence time. Field experience has shown that residence time is one of the critical parameters influencing the treatment performance of stormwater constructed wetlands (e.g., Walker, 1998, 2001; Werner and Kadlec, 1996; Somes et al., 2000).

Tilley and Brown (1998) summarizing the results from stormwater wetlands in south Florida reported that at the neighborhood scale (10-100 ha) phosphorus runoff required the largest wetland treatment area, needing between 2.3 and 10.8% of the total basin area. At the sub-basin scale (100-1,000 ha) the loading of TSS, derived from land use specific criteria, needed the largest treatment area, ranging from 0.2 to 4.5% of basin area. The basin scale (> 1,000 ha) treatment, based on retaining drainage canal discharge for at least 72 hours, needed between 0.1 and 2.5% of basin area.

Table 4-13. Examples of stromwater runoff FWS constructed wetlands with emergent vegetation.(Pesticide containing runoff in Table 4-15)

Type of runoff	Location	Size (A) (ha)	Catchment (B)(ha)	A:B (%)	Target pollutant	Ref.
Dairy pasture	Wodonga, Victoria, Australia	0.045	90	0.05	Nutrients	1
Dairy pasture	Kiwitahi, New Zealand	0.026	2.6	1.0	Nutrients	2
Dairy pasture	Crooke´s wetland, NSW, Australia	0.036	90	0.004	Nutrients	3
Agric. tile drainage	Champaign County, IL, USA	0.6; 0.8	15; 25	4; 3.2	Nutrients	4
Agricultural drainage	Maine, USA	0.61	7	8.7	Nutrients, TSS	5
Agricultural drainage, irrigation	Legnaro, Italy	0.32	6	3.3	Nutrients	6
Agricultural runoff	Everglades, FL, USA	1,546			Phosphorus	7
Agricultural runoff (sugarcane)	Nanga Farms, Zambia	5.52	1,539	0.36	Ogranics, nutrients	8
Agricultural irrigation runoff	Corcoran, CA, USA	1.14			Se	9
Residential catchment	Plumpton Park, NSW, Australia	0.45	75	0.6	Nutrients, TSS, bacteria	10
Residential catchment	Woodcroft Estate, NSW, Australia	1.5	53	2.8	Nutrients, TSS, bacteria	10
Residential catchment	Brisbane, QNS, Australia	0.8	180	0.45	TSS, nutrients	11,12,13
75% residential/25% agricultural	Fremont, CA, USA	20	1,200	1.7	Heavy metals, nutrients	14
Residential/industrial catchment	Barker Inlet (Adelaide), Australia	11*	2,050	0.54	Heavy metals	15
Road runoff	Dagenham, United Kingdom	0.175			Cd, Pb	16,17
Road runoff	Newbury, United Kingdom	0.0185	1.6	1.15	Organics, heavy metals	18
Road runoff	Newbury, United Kingdom	0.54	3.4	15.9	Heavy metals,	19
Road runoff	Kildare, Ireland	0.0273			TSS, organics, h. metals	20
70% urban/30% forest runoff	Croudace Bay, NSW, Australia		68		TSS, nutrients, bacteria	21
Urban/agricultural runoff	Kanata, Ontario, Canada	3.13	637	0.02	Fe, Mn	22
42%runoff/58% wastewater	Aiken, SC, USA	77	3.22	4.2	Cu	23
Airport de-icing runoff	Kalmar, Sweden	18			Nitrogen	24

1-Raisin et al. (1997), 2-Tanner et al. (2005b), 3- Raisin and Mitchell (1994), 4-Larson et al. (2000), 5-Higgins et al. (1993), 6-Borin et al. (2001), 7-Nungesser and Chimney (2001) treats $5.3 \times 10^6\ m^3\ yr^{-1}$, 8-Musiwa et al. (2002), 9-Gao et al. (2003), 10-Davies et al. (2001), Bavor et al. (2001), 11-Kasper and Jenkins (2005), 12- Bayley and Greenway (2005), 13-Greenway et al. (2006), 14-Meiorin (1989), 15-Walker and Hurl (2002) *part of a 1.27 km^2 wetland system, 16-Carapeto and Purchase (2000), 17-Scholes et al. (1999), 18-Pontier et al. (2004), 19-Hares and Ward (2004), Silverman (1989), 120-Higgins et al. (2006), 21-Geary et al. (2006), 22-Goulet and Pick (2001), 23-Murray-Gulde et al. (2005a,b), 24-Thorén et al. (2003)

Figure 4-27. FWC constructed wetlands for golfcourse runoff in Rivertowne, South Carolina, USA (top left, dominant plant *Sagittaria latifolia*), parking lot runoff in Charleston, South Carolina (top right, *Typha latifolia*), highway intersection runoff in Boucherville near Montréal, Québec, Canada (middle left, *Phragmites australis*), highway runoff in West Palm Beach, Florida, USA (middle right, *Thalia geniculata*), stormwater runoff from residential area in Plumpton Park, NSW, Australia (bottom left, *Phragmites australis*) and in Bonnyrigg Park, Fairfield, NWS, Australia (bottom right, *Typha* spp. *Baumea articulata*). Photos by Jan Vymazal.

Figure 4-28. FWS constructed wetland (2.3 ha, in a white circle) treating runoff waters from 40 ha pasture catchment before discharging into Lake Okaro, New Zealand. Top - general view after construction, photo provided by Environment Bay of Plenty, New Zealand, bottom: details of vegetation: *Scirpus tabernaemontani* (left) and *Bolboschoenus fluviatilis* (right). Photo Jan Vymazal.

Figure 4-29. FWS constructed wetland (90 m^2) for removal of nitrate from a 1.7 ha pasture at Bog Burn, New Zealand (South Island). Photo by Chris Tanner, with permission.

4.1.4.5 Mine drainage

A very large application area for constructed wetlands is the treatment of acid coal mine drainage (Wieder, 1989; Ziemkiewicz et al., 2003). The contaminants of interest are typically pH, sulfate, aluminum, iron and manganese (Table 4-14, Figs. 4-30 and 4-31). Despite the large number of such wetlands, no clearly stated design methodology is yet available for acid mine drainage (Kadlec et al., 2000). However, Tarutis et al. (1999) reported that treatment efficiency is linearly related to area adjusted removal and exponentially related to first-order removal at constant hydraulic loading rates. If removal is zero-order, the wetland area required to treat a discharge to meet some target effluent concentration is a decreasing linear function of influent concentration. However, if removal is first-order, the required wetland area is a non-linear function of the relative influent concentration.

Early constructed wetlands were built to mimic the peat (*Sphagnum*) wetlands that first showed that the quality of mine water was improved as it passed through these wetlands (e.g., Burris, 1994; Kleinmann, 1985; Girts and Kleinmann, 1986). However, *Sphagnum* wetlands were difficult to establish and maintain, and the design was replaced by one in which emergent plants, most often cattails (*Typha* spp.) due to their ability to survive in low pH-waters, are dominant vegetation (Girts et al., 1987; Kadlec et al., 2000).

The treatment of mine drainage by wetlands has evolved from simple FWS systems to sequential treatment in variety of wet environments. Recently, passive treatment options have been expanded to include anoxic limestone drains, which add alkalinity to the drainage before treatment wetland (e.g. Nairn and Hedin, 1992; Brodie et al., 1993). Metals precipitate in aerated water as oxides, hydroxides and oxyhydroxides (see section 2.6). Sulfate may be reduced in anaerobic zones of the wetland, but in general, FWS constructed wetlands do not provide optimal conditions for sulfate reduction (Sheoran and Sheoran, 2006).

A small but growing application are for constructed wetlands is the treatment of various metal-mine drainage waters and tailing pond waters (Table 4-14, Fig. 4-32). Metals are removed by cation exchange to wetland sediments, precipitation as sulfides and other insoluble salts, and plant uptake. As sulfide reduction is limited in FWS wetlands, probably the major removal mechanism is co-precipitation with ferric and manganic hydroxides and oxyhydroxides (Sheoran and Sheoran, 2006).

Table 4-14. Examples of the use of FWS constructed wetlands to treat mine drainage waters.

Type of drainage	Location	Target pollutants	Ref.
Ironstone spoil heap (acid)	Scotland	pH, Fe, Mn, Al, SO_4^{2-}	1
Spoil heaps (neutral)	England	Fe, Mn, Al, SO_4^{2-}	2
Acid coal mine	Kentucky, USA	Fe, Al, Mn, acidity	3
Acid coal mine	Tennessee, USA	pH, Fe, Mn	4
Acid coal mine	Ohio, USA	Al, Fe, Mn, SO_4^{2-}	5
Acid coal mine	Spain	pH, Fe. Mn, SO_4^{2-}	6
Acid coal mine	Kentucky, USA	Fe, Al, Mn, acidity	7
Acid coal mine	South Africa	pH, Fe, Mn, SO_4^{2-}	8
Acid lignite mine	Spain	pH, Fe, Al	9
Acid/alkaline coal mine	USA	pH, Fe, Al, Mn, SO_4^{2-}	10
Coal ash disposal area	Alabama, USA	pH, Fe, Mn, heavy metals	11
Surface coal mine seep	Ohio, USA	Fe, Mn	12
Coal slurry	Alabama, USA	Mn, Fe, pH, TSS	13
Acid coal mine and U, Cu, Zn tailing s	Ontario, Nova Scotia, Canada	Cu, Zn, pH, U, Fe	14
Uranium mine	NT, Australia	Uranium	15
Uranium mine	Germany	Uranium	16
Lead-zinc mine spent water	Ireland	Zn, Pb, SO_4^{2-}	17
Copper mine	B.C., Canada	Cu	18
Mine stockpiles	Minnesota, USA	pH, Cu, Ni, Co, Zn	19
Gold mine	Ontario, Canada	Nitrogen, cyanide	20
Heavy metals mine	Cornwall, UK	Heavy metals	21, 22

1-Woulds and Ngwenya (2004), 2-Batty et al. (2005), 3-Karathanasis and Johnson (2003), 4-Brodie et al. (1988), 5-Mitsch and Wise (1998), 6-Ramírez Masferrer (2002), 7-Barton and Karathanasis (1999), 8-Wood and Cook (1992), 9-de Matos and da Gama (2004), 10-Hedin and Nairn (1993), 11-Mays and Edwards (2001), 12-Stillings et al. (1988), 13-Brodie et al. (1986), 14-Kalin et al. (1989), 15-Overall and Parry (2004), 16-Kiessig et al. (2003), 17-O'Sullivan et al. (2004), 18-Sobolewski (1996), 19-Eger et al. (1993), 20-Bishay and Kadlec (2005), 21-Whitehead and Prior (2005), 22-Hallberg and Johnson (2005)

Figure 4-30. Treatment of abandoned mine drainage in Monastery Run, Pennsylvania, USA. Photo by Jan Vymazal.

Figure 4-31. FWS constructed wetland for acid mine drainage in Lick Run, Ohio, USA. For details on performance see Mitsch and Wise (1998). Photo by Jan Vymazal.

Figure 4-32. FWS constructed wetland planted with *Eleocharis dulcis* for the treatment of mine waters from Ranger uranium mine at Jabiru, Northern Territory, Australia. Photo by David Jones, with permission.

4.1.4.6 Other uses of FWS systems

Besides municipal and animal wastewaters, stormwater runoff and mine drainage waters, which are described above, constructed wetlands with free water surface and emergent vegetation have also been used for a wide variety of other types of wastewater including pulp and paper production (Fig. 4-33), landfill leachate (Fig. 4-34), petroleum industry, food processing wastewaters (Figs. 4-35 and 4-36), fish and shrimp aquaculture water, pesticide containing runoff or explosives. Examples of the use FWS constructed wetlands for various types of wastewater are presented in Table 4-15.

Table 4-15. Examples of the use of FWS constructed wetlands with emergent vegetation for the treatment of various types of pollution.

Wastewater	Location	Reference
Refinery process water	USA China Hungary	Litchfield and Schatz (1989), Litchfield (1993), Gillepsie et al. (2000), Dong and Lin (1994) Lakatos (1998)
Pulp and paper industry	USA China	Hatano et al. (1992), Moore et al. (1992), Tettleton et al. (1993), Knight et al. (1994) Xianfa and Chuncai (1994),
Pesticides	USA South Africa Norway	Alvord and Kaldec (1996), Runes et al. (2003), Moore et al. (2000b, 2001, 2002), Weaver et al. (2004), Sherrard et al. (2004), Schulz and Peall (2001) Braskerud and Haarstad (2003)
Electric utility WW	USA	Ye et al. (2003)
Fish pond effluent	USA	Schwartz and Boyd (1995)
Shrimp aquaculture	USA	Tilley et al. (2002)
Salmon hatchery	USA	Michael (2003)
Abattoir facility	Canada	Goulet and Sérodes (2000)
Landfill leachate	Sweden Norway Canada USA	Benyamine et al. (2004) Mæhlum (1994) Sartaj et al. (1999) Martin et al. (1993), Johnson et al. (1999)
Tool industry	Aregntina	Hadad et al. (2006)
Metals-tertiary	USA	Dombeck et al. (1998)
Power station processwater	Australia	Jensen et al. (2006)
Sugar factory	Kenya	Tonderski et al. (2005), Bojcevska et al. (2006)
Olive mill	Greece	Kapellakis et al. (2004)
Sugar beet process	USA	Anderson (1993)
Potato processing	USA	Kadlec et al. (1997)
Explosives	USA	Best et al. (2000)
Hydrocarbons	France	Salmon et al. (1998)
Woodwaste leachate	Canada	Tao and Hall (2004), Masbough et al. (2005)
Soft drink industry	Uruguay	Perdomo (pers. comm.)

Figure 4-33. FWS constructed wetland for secondary treatment effluent from bleached kraft mill in Pensacola, Florida. (Details on performance in Knight et al., 1994). Photo by Jan Vymazal.

Figure 4-34. FWS constructed wetland for municipal landfill leachate at the Perdido Landfill near Pensacola, Florida. Detail of one out of 10 cells (91 x 11 m each) connected in series. For details on performance see DeBusk (1999) or Martin and Moshiri (1992). Photo by Jan Vymazal.

Figure 4-35. FWS constructed wetlands for abattoir wastewater treatment at Pouliot, Québec, Canada. Photo by Jan Vymazal.

Figure 4-36. FWS constructed wetland planted with *Typha domingensis* for treatment of soft drink enterprise wastewaters NATIVA II in Uruguay. Photo by Jan Vymazal.

4.1.5 Systems floating mats of emergent plants

Some emergent macrophytes are capable of forming floating mats (Fig. 4-37), even though their individual plants are not capable of such existence. *Typha* spp. (cattails), *Glyceria maxima* (Giant sweetgrass, Mannagrass), *Phragmites australis* (Common reed), *Cyperus papyrus* (Papyrus) or *Alternanthera philoxeroides* (Alligator weed) are examples of plants which all capable of growing in mats. Treatment systems have been operated in this fashion; however, in some cases the floating mats developed unintentionally (Kadlec and Bevis, 1990; Kalin and Smith 1992).

Kadlec and Bevis (1990) reported that the *Typha* mat is stable as long as it retains sufficient areal extent. If small portions of the "raft" detach, the plants are top-heavy and tip over. Therefore, systems which are intentionally designed with floating mats use some kind of structures which keep the plant in upright position.

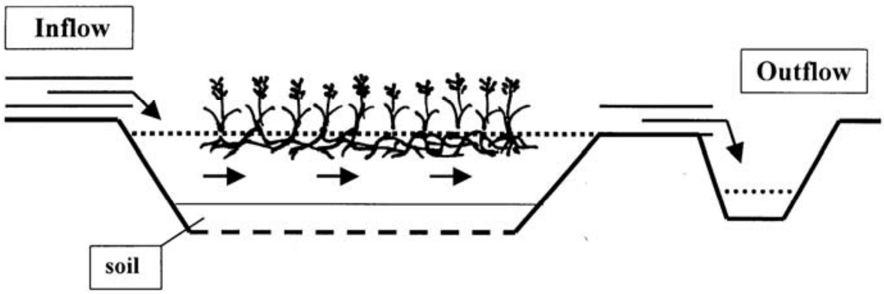


Figure 4-37. Schematic representation of a constructed wetland with floating mats of emergent macrophytes. From Vymazal (2001a) with permission from Backhuys Publishers.

Van Ooostrom and Russell (1992) reported the use of system with floating *Glyceria maxima* for removal of nitrogen, especially nitrate from a nitrified meat processing effluent in New Zealand. Sections of mat of *G. maxima* were cut from a mature wetland and floated on the wastewater to rapidly establish a plant cover on the wetland. No soil or sediment was placed in the bottom of the system. The NO_x-N removal rate in the wetland averaged 548 g m^{-2} yr^{-1} (1.5 g m^{-2} d^{-1}) being 83% of the total nitrogen removal. The active denitrification sites in the floating wetland system were the plant mat and the sediments (which developed during the operation of the system), both of which supply organic carbon and anaerobic sites for denitrification. van Oostrom (1995) reported that nitrogen removal in wetlands with floating mats of *Glyceria maxima* during the final year of the study was about twice that in the wetland without plants, and averaged 46-49% (1 898 – 2 008 g N m^{-2} yr^{-1}) at a high loading rate of 4 125 g N m^{-2} yr^{-1}.

Van Bruggen et al. (1994) found that the mass balance for nitrogen removal in an experimental wetland unit with floating *Cyperus papyrus* indicated that 81% of the nitrogen loaded in the wetland was taken up by the plants, whereas 14% could be found in the outflow. In the units without plants, 38% of the nitrogen load was found in the effluent and 59% is unaccounted for, probably due to denitrification. Also Okurut et al. (1999) reported that FWS wetland in Uganda with floating mats of *Cyperus papyrus* and *Phragmites mauritianus*. The planted units were more efficient that unplanted unit in removal of COD, BOD_5 and phosphate while removal of ammonia was higher in unplanted systems due to more anoxic/anaerobic conditions in the water under the *Cyperus* and *Phragmites* mats. Okurut (2001) reported that growing plants in floating mats give the advantage of allowing the harvesting also the roots and rhizomes and thus removing more nutrients as compared to aboveground biomass only.

Smith and Kalin (2002) reported that since 1990, floating rafts of *Typha* sp. were grown in open mine pits flooded with pH 6 mine drainage (central Newfoundland), a biological treatment system for acid (pH 3.5) mine drainage (northern Ontario) and on a settling pond for particulates in coke stockpile runoff (northern coastal British Columbia) to remove suspended solids. A root surface area of 15 m^2 per m^2 of mat, harboring 0.3 kg of particulates per m^2 was determined after the first growing season. Root surface areas of 114 m^2 per m^2 of mat were measured in five year-old populations.

In 2002, London Heathrow International Airport completed a construction of a 10 ha treatment system designed primarily to treat the de-icing contaminant runoff from an extensive catchment of some 600 ha of runway, taxiways, cargo areas and terminal buildings. The system comprises a series of aerated balancing ponds combined with over 2 ha of sub-surface flow constructed wetlands, together with a kilometer of rafted reed beds (Worall, 2006, Fig. 4-38). In 2000, FWS constructed wetland with floating mats of emergent macropgytes was put in operation in Bornem, Belgium, to treat combined sewer overflow (Van de Moortel et al., 2006, Fig. 4-39).

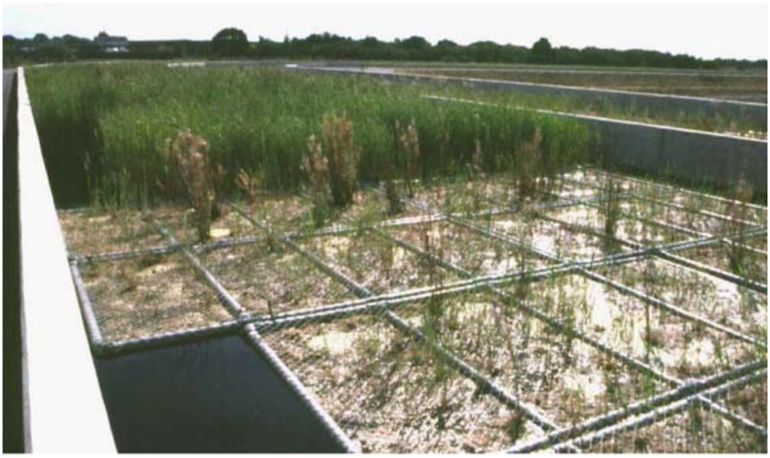


Figure 4-38. FWS constructed wetland with floating rafts with *Phragmites australis* at London Heathrow International Airport treatment system. Photo by Jan Vymazal.

Figure 4-39. FWS constructed wetland with floating mats of emergent macrophytes (e.g., *Sparganium, Juncus, Epilobium, Scirpus*) for treatment of combined sewer overflow in Bornerm, Belgium. Plants are kept upright using coconut string grid. Photo by Lenka Kröpfelová.

Kyambadde et al. (2005) reported on the use of two tropical emergent macrophytes *Cyperus papyrus* and *Miscanthidium violaceum* in a floating mode in constructed wetlands in Uganda. The results showed that vegetated units, and especially one planted with *Cyperus papyrus*, exhibited high ability to treat high oxygen demanding and nutrient rich wastewater.

4.2 Sub-surface systems

Constructed wetlands with sub-surface flow may be classified according to the direction of flow into horizontal (HF or HSF) and vertical (VF) (Fig. 4-1). Vertical-flow treatment wetlands could be further categorized into down-flow and up-flow whether the wastewater is fed onto the surface or to the bottom of the wetland (Fig. 4-1).

4.2.1 Horizontal flow

This section contains only a brief description of horizontal sub-surface flow (HF) constructed wetlands. More detailed information on this treatment technology, treatment performance, types of wastewater treated in HF constructed wetlands and evaluation of HF constructed wetlands use around the world is included in Chapter 5.

Figure 4-40 shows a typical arrangement for the constructed wetland with a horizontal sub-surface flow. It is called horizontal flow because the wastewater is fed in at the inlet and flows slowly through the porous medium under the surface of the bed in a more or less horizontal path until it reaches the outlet zone where it is collected before leaving via level control arrangement at the outlet. During this passage the wastewater will come into contact with a network of aerobic, anoxic and anaerobic zones. The aerobic zones occur around roots and rhizomes that leak oxygen into the substrate (Brix, 1987b; Cooper et al., 1996). In Europe, HSF constructed wetlands are commonly called “Reed beds”, in the United Kingdom also “Reed Bed Treatment System” (RBTS) coming from the fact that the most frequently used plant is Common reed (*Phragmites australis*). In the United States, the term “Vegetated Submerged Bed” (VSB) was also adopted. This term, however, is very unfortunate as it resembles systems with submerged plants and therefore, we strongly recommend that it is not used.

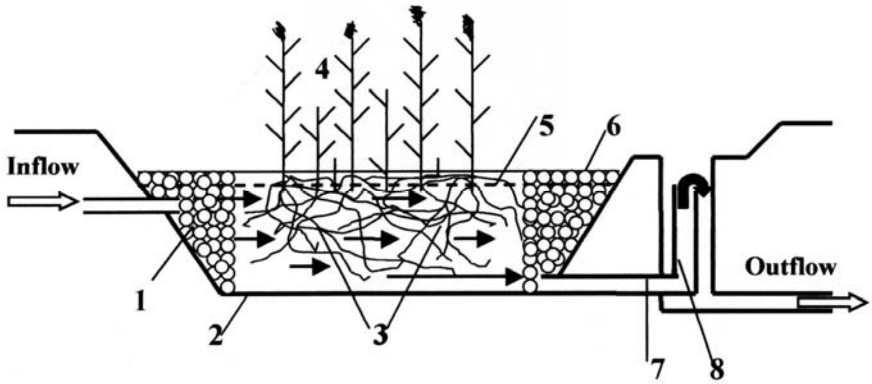


Figure 4-40. Schematic representation of a constructed wetland with horizontal sub-surface flow. 1 - distribution zone filled with large stones, 2 – surface of the bed, 3 – water level in the bed, 4 - impermeable liner, 5 - medium (e.g., gravel, crushed stones), 6 –collection zone filled with large stones, 7 – collection drainage pipe, 8 - outlet structure for maintaining of water level in the bed. The arrows indicate only a general flow pattern. From Vymazal (2001a) with permission from Backhuys Publishers.

Organic compounds are degraded aerobically as well as anaerobically by bacteria attached to the plant´s underground organs (i.e. roots and rhizomes) and media surface. The oxygen required for aerobic degradation is supplied directly from the atmosphere by diffusion or oxygen leakage from the macrophyte roots and rhizomes in the rhizosphere (sections 2.1 and 3.2.1).

Numerous investigations have shown that the oxygen transport capacity of the reeds is insufficient to ensure aerobic decomposition in the rhizosphere and that anoxic and anaerobic decomposition play an important role in HF constructed wetlands (Brix, 1990a; Vymazal and Kröpfelová, 2006).

The existence of a subsurface air/water interface causes sediment processing in the HF wetland to differ considerably from that in FWS wetlands. Macrophyte aboveground biomass and litter are mostly contained on the surface of the bed and do not interact with the water flowing in the interstices below. However, many particulate processes do operate in the water-filled voids. Particles settle into stagnant micropockets or are filtered out by flow constrictions (Kadlec et al., 2000).

The major removal mechanism for nitrogen in HF CWs are nitrification/denitrification reactions (Vymazal 1999c, Vymazal, 2007). However, the field measurements have shown that the oxygenation of the rhizosphere of HF constructed wetlands is insufficient and, therefore, the incomplete nitrification is the major cause of limited nitrogen removal (Brix and Schierup, 1990; Vymazal 2007). Volatilization, adsorption and plant uptake play much less important role in nitrogen removal in HF constructed wetlands (Cooper et al., 1996; Vymazal, 1999c, Vymazal et al., 1998a). Volatilization is limited by the fact that HF CWs do not have free water surface and algal activity is negligible in these systems. The fine-grained soils always show better nitrogen removal through adsorption than the coarse-grained soil (Geller et al., 1990). The higher elimination rate can be explained by the higher cation exchange capacity of the fine-grained soils. However, fine-grained soils are not used for HF systems, at present, because of poor hydraulic conductivity. Therefore, the adsorption capacity of the commonly used media (pea gravel, crushed rock) is very limited. Nitrogen removal via plant uptake and subsequent harvesting is usually low as compared to inflow loads (e.g. Vymazal, 2005a).

Phosphorus is removed primarily by ligand exchange reactions, where phosphate displaces water or hydroxyls from the surface of Fe and Al hydrous oxides. However, media used for HF wetlands (e.g., pea gravel, crushed stones) usually do not contain great quantities of Fe, Al or Ca and therefore, removal of phosphorus is generally low. It has been found that removal of nitrogen and phosphorus through plant harvesting is negligible and forms only a tiny fraction of the amount removed. Plant uptake removal mechanisms are limited in temperate and colder regions because of harvesting regime which does not allow harvesting of macrophytes, and especially *Phragmites australis*, during the peak nutrient standing stock in the late summer. However, this mechanism may play more significant role in nutrient removal in tropical and partially subtropical regions where the growth seasonality and nutrient translocations between above- and below-ground parts are minimal.

Microbial pollution removal is mainly achieved through a combination of physical, chemical and bilogical factors (Pundsack et al., 2001; Vymazal, 2005d). Physical factors include mainly filtration and adsorption (e.g., Stevik et al., 2004). Chemical and biological factors may include oxidation, exposure to biocides excreted by a number of wetland plants, antimicrobial activity of root exudates, predation by nematodes and protists, activity of lytic bacteria or viruses, retention in biofilms and natural die-off (Kickuth and Kaitzis, 1975; Seidel, 1976; Gersberg et al., 1989 a,b; Hatano et al., 1993; Brix, 1997; Decamp and Warren, 1998; Decamp et al., 1999; Neori et al., 2000).

4.2.2 Vertical flow

4.2.2.1 Downflow

The earliest form of vertical flow (VF) system is that of Seidel in Germany in the 1970s, sometimes called the Max Planck Institute Process (MPIP) or the Krefeld Process (Seidel, 1978). Similar systems in the Netherlands were called "infiltration fields" (Greiner and de Jong, 1984). Interest in the particular process seemed to wane but it has been revived in the last decade because of the need to produce beds which nitrify. HF beds have low ability to oxidize ammonia to nitrate mainly due to insufficient amount of oxygen transferred by macrophytes to the rhizosphere Cooper et al. 1996). Typical arrangement of downflow VF system is shown in Fig. 4-41.

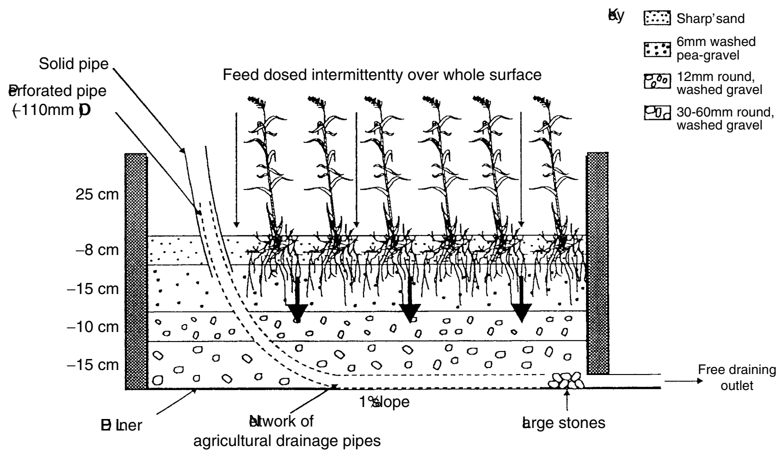


Figure. 4-41. Typical arrangement of a downflow vertical-flow constructed wetland (from Cooper et al., 1996, with permission from Water Reseach Centre plc).

Vertical flow (VF) constructed wetlands comprise a flat bed of graded gravel topped with sand planted with macrophytes. The size fraction of gravel is larger in the bottom layer (e.g., 30-60 mm) and smaller in the top layer (e.g., 6 mm). VF CWs are fed intermittently with a large batch thus flooding the surface. Wastewater then gradually percolates down through the bed and is collected by a drainage network at the base. The bed drains completely free and it allows air to refill the bed. This kind of dosing leads to good oxygen transfer and hence the ability to nitrify (Cooper et al., 1996). The oxygen diffusion from the air contributes much more to the filtration bed oxygenation as compared to oxygen transfer through plants' aerenchyma system. The major purpose of macrophyte presence in VF CWs is to help maintain the hydraulic conductivity of the bed. Many of the vertical-flow systems are staged systems with parallel and series beds. There are sometimes parallel first-stage beds which are fed in rotation for 1 to 2 days and then rested for periods of 3 to 6 days (Cooper et al., 1996).

Cooper (2005) pointed out that the most important factors to achieve in the design of a VF are:

1. To produce a bed matrix that allows the passage of the wastewater through the bed before the next dose arrives whilst at the same time holding the liquid back long enough to allow the contact with the bacteria growing on the media and achieve the required treatment.

2. To provide sufficient surface area to allow the oxygen transfer to take place and sufficient bacteria to grow.

The early VF systems have usually been composed of several stages with 2-4 beds in the first stage which were fed with wastewater in rotation (Fig. 4-42). Such VF systems are now called <u>1st generation VF systems</u> (Cooper, 2005). The early VF systems were frequently the first stage of the hybrid systems (Burka and Lawrence, 1990; Liénard et al., 1990, 1998; Cooper et al., 1996; Mitterer-Reichmann, 2002). Recently, VF systems with only one bed have been used (Fig. 4-43). These systems are called <u>2nd generation VF constructed wetlands</u> or <u>compact vertical flow beds</u> (e.g., Weedon, 2003; Mitterer-Reichmann, 2002; Brix et al., 2002; Johansen et al., 2002; Arias and Brix, 2005a).

In France, more than 1,000 VF systems treat unsettled (raw) sewage (Boutin and Liénard, 2003; Molle et al., 2005a; Paing and Voisin, 2005; Paing et al., 2006; Esser et al., 2007). The special fewture of this system is that it accepts raw sewage directly onto the first stage allowing for easier sludge management in comparison to dealing with primary sludge from an Imhoff tank (Molle et al., 2005a). The most commonly used format is that a series of first stage beds dosed in rotation acts as combination of a sludge drying reed bed and pretreatment and the second stage is a "conventional" VF bed. While compact VF systems are usually small (Fig. 4-43), VF systems treating raw sewage in France are often much larger (Fig. 4-44).

Figure 4-42. Hybrid system at Oaklands Park, UK. The first VF stage (6 x 8 m^2) is composed of six beds planted with *Phragmites australis* (top back of the picture), the second VF stage (below, 3 x 5 m^2) is composed of three beds planted with *Scirpus lacustris*, *Iris pseudacorus* and *P. australis*. The third HF stage (8 m^2) is planted with *I. pseudacorus* and the 4th HF stage (20 m^2) is planted with *Acorus calamus*, *S. lacustris*, *Carex elata* and *Sparganium erectum*. Photo by Jan Vymazal.

Figure 4-43. A single bed VF constructed wetland (15 m^2) at Mørke, Denmark, planted with *Phragmites australis*. Left: shortly after planting in 2002 (from Arias and Brix, 2005a, with permission from Backhuys Publishers), Right: in June 2005, photo by Jan Vymazal.

Figure 4-44. VF constructed wetlands treating raw sewage from 1,200 PE in Roussilon, France. Photo by Jan Vymazal.

Various parameters are used to calculate the necessary area of VF constructed wetlands. Cooper (2005) summarized the most frequently used equations using the following nomenclature:

A = area of bed needed
A_1 = area of stage 1
A_2 = area of stage 2
P = population equivalent

Grant (1995) for population up to 100 PE:

$A_1 = 3.5\ P^{0.35} + 0.6\ P$ (4.5)
$A_2 = 0.5A_1$ (if septic tank used) (4.6)
$A_2 = 0.6A_1$ (if no septic tank used) (4.7)

Cooper et al. (1996) "Rule of thumb":

A = 1.0 P (BOD removal only) (4.8)
A = 2.0 P (BOD and NH_4-N removal, split in 2 stages) (4.9)

Weedon (2001):

$A = 5.4\ P^{0.6}$ (for populations up to 25 PE) (4.10)

$A = 2.4\ P^{0.85}$ (for populations greater than 25 PE) (4.11)

Grant and Griggs (2001):

$A = 5.25\ P^{0.35} + 0.9\ P$ (updated version of Grant (1995)) (4.12)

Platzer (1999) – area based on oxygen transfer rate* of 28 g O_2 m^{-2} d^{-1}

Boutin and Lienard (2003):

$A = 2.5P$ (split in two stages) (4.13)

Recently, other design equations have been proposed in France (Molle et al., 2005a), Denmark (Brix and Johansen, 2004), Austria (Langergraber et al., 2006; ÖNORM, 2005) or Germany (Fehr et al., 2003, DWA, 2006):

Molle et al. (2005a):

$A = 2.0\ P$ (4.14)
(split in 1.2 P and 0.8P for first and second, stages, respectively)

According to the French experience, two-stage VF systems designed according to this equations and treating raw sewage should produce an outflow COD concentrations < 60 mg l^{-1}.

Brix and Johansen (2004):
$A = 3.0\ P$ (4.15)

VF systems designed with the area of 3 m^2 per one population equivalent should be able to fulfill 95% reduction of BOD_5, provided that effective depth of the filter is 1 m and the filter medium is sand with a d_{10} between 0.25 and 1.2 mm and d_{60} between 1 and 4 mm.

Langergraber et al. (2006), ÖNORM (2005), DWA (2006), Fehr et al. (2003):

$A= 4.0\ P$ (4.16)

The Austrian design is based on the fact that VF constructed wetlands with intermittent feeding operated with an organic load of 20 g COD m^{-2} d (i.e., 4 m^2 per person) can fulfill the requirements of the Austrian standard regarding maximum effluent concentrations and minimum elimination efficiencies. Similar results have been reported from Germany.

Table 4-16 shows the comparison between the area needed for VF systems resulting from using the equations (4.5) to (4.13) for two sizes of group, a family of 4 people and hamlet of 50 people using 60 g BOD_5, 12 g NH_4-N and 0.2 m^3 per population equivalent.

One of the major threats of good performance of VF systems is clogging of the filtration substrate (Platzer and Mauch, 1997; Langergraber et al., 2003; Winter and Goetz, 2003; Chazarenc and Merlin, 2005). Therefore, it is important to properly select the filtration material, hydraulic loading rate and distribute the water evenly across the bed surface.

In the UK, layers of graded material are used (Fig. 4-41). In France, VF systems treating raw sewage use graded gravel layers from top to the bottom (2-8 mm, 10-20 mm and 20-40 mm) in the first stage and layers of sand (0-2 mm) and gravel (3-8 mm and 10-20 mm) in the second stage (Paing et al., 2006). More details on filtration materials in VF systems are given e.g. by Arias et al. (2003a), Molle et al. (2005a).

Hydraulic loading rates (HLR) reported in the literature vary greatly. Cooper et al. (1997) reported an average HLR of 30.6 cm d^{-1} with the peak HLR of 150 cm d^{-1} during the dosing period. Mitterer-Reichmann (2002) reported an average HLR for 200 VF system in Austria of 2.7 cm d^{-1} on 5.5 m^2 PE^{-1}. Langergraber et al. (2003) did not observe clogging at less than 10 cm d^{-1}. Weedon (2001, 2003) reported that 2nd generation VF beds worked without flooding in the range 3.3 to 102.7 cm d^{-1} but that it flooded at 127 cm d^{-1}. Johansen et al. (2202) operated another 2nd generation system in Denmark over the range 20 to 120 cm d^{-1} with no flooding below 80 cm d^{-1}.

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*Cooper (1999) pointed out that the sizing of the beds is intimately linked to the oxygen transfer capacity of the system. This is linked to the intermittent dosing system used and the hydraulic loading rate. A crude estimation of the necessary total oxygen transfer rate (OTR) across vertical-flow stage was presented by Cooper (1999):

$$\frac{[(BOD_{in} - BOD_{out}) + (NH_4^+\text{-}N_{in} - NH_4^+\text{-}N_{out}) \times 4.3\,] \times \text{flowrate/day}}{\text{area of bed}} \tag{4.17}$$

This approximation does not allow for (Cooper 1999):

- BOD removal by settlement/filtration
- BOD removal by denitrification
- NH_4^+-N lost through a) plant uptake, b) volatilization or c) adsorption

The value 4.3 used to estimate the O_2 needed for ammonia oxidation comes from work on nitrification in activated sludge systems. The author pointed out that NH_4^+-N for a fresh sewage is also an underestimate of the potential NH_4^+-N when all the organically-bound nitrogen in urea has been hydrolyzed. Therefore, TKN is a better measure but is not often reported.

Platzer (1998, 1999) uses an equation similar to equation (4.14) where (BOD_{in} - BOD_{out}) is replaced by 0.7 (COD_{in} - COD_{out}). Platzer (1998, 1999) recommends a maximum limit of 25 g COD m^{-2} d^{-1} or 6.5 g TKN m^{-2} d^{-1} for designing VF systems. This is virtually the same value of since it takes 4.3 g O_2 to oxidize 1 g TKN and this would be 28 g O_2 m^{-2} d^{-1} (Cooper, 2005).

Table 4-16. Comparison of the area needed (m^2 per PE) for VF beds using the various equations (Cooper, 2005). Reprinted from Water Science and Technology, Volume 51, Issue 9, pp.81-90 with permission from the copyright holder, IWA.

Authors	**BOD_5 and NH_4-N oxidation**	
	for 4 PE	**for 50 PE**
Grant (1995)	2.0	0.88
Grant and Griggs (2001)	3.0	1.31
Cooper et al. (1996)	2.0	2.0
Weedon (2001, 2003)	3.1	1.33
Platzer (1999)	0.3	0.3
Brix et al. (2002), Brix (2003a)	0.9	1.50
Boutin and Liénard (2003)	2.0-2.5*	2.0-2.5*
Cooper and Cooper (2005)	2.5	2.5

*Area needed depends upon whether or not the sewage as from separate or combined system.

It is also important to distribute the wastewater evenly across the surface of VF bed in order to avoid overloading of certain parts of the surface. The early systems used a network of pipes with downward pointing holes (Fig. 4-45). At present, wastewater distribution system mostly consists of perforated polyethylene pipes laid down on the bed surface (Fig. 4-46).

The major treatment processes in VF beds are the same as in HSF CWs. However, VF beds are far more aerobic then HF beds and are good for nitrification as well as BOD removal. On the other hand, VF beds do not provide much denitrification. They are less good than HSF beds for suspended solids removal. Removal of phosphorus depends on the use of filtration media which can chemically bind the phosphate to the media. A wide variety of materials have been studied including apatite (Molle et al. (2005b), calcite (Arias et al., 2003b), light weight aggregate (LWA) (Zhu et al., 2003; Adám et al., 2005), blast furnace granulated slag (Grüneberg and Kern, 2001; Korkusuz et al., 2004, 2006) or iron ore (Grüneberg and Kern, 2001).

Treatment performance of downflow VF constructed wetlands is summarized in Table 4-17. Summary of VF systems performance in France has been summarized by Molle et al. (2005a) or Paing et al. (2006), Austrian experience was summarized by Miterrer-Reichamnn (2002) and Danish experience has been summarized by Brix and Johansen (2004).

VF constructed wetlands are efficient in removal of organics (BOD_5 and COD) and suspended solids. Removal of ammonia-N is high as compared to other types of constructed wetlands due to good oxygenation of the filtration bed as a consequence of intermittent feeding. However, removal of total nitrogen is comparable with FWS and HF systems due to inability to provide denitrification. This could be resolved by recycling of the effluent into the pretreatment unit, e.g. septic or Imhoff tank (Arias and Brix, 2006). Removal

Figure 4-45. Wastewater distribution at vertical flow constructed wetland which is a part of the treatment system for landfill leachate near Leiria, Portugal. Photo by Jan Vymazal.

Figure 4-46. Distribution of wastewater across the surface of VF bed at Bojna, Slovakia. Photo by Milan Raffesberg, with permission.

of phosphorus is also comparable with other types of constructed wetlands. Phosphorus removal could be enhanced by the use of filtration materials with high sorption capacity (as discussed above).

Table 4-17. Treatment performance of downflow vertical-flow constructed wetlands: world-wide experience. In = inflow, Out = outflow, Eff = efficiency, Rem = removed load, n = number of systems. *FC = fecal coliforms (log CFU 100 ml^{-1}).

	Concentration (mg l^{-1})				Loading (kg ha^{-1} d^{-1})			
	In	Out	Eff.(%)	n	In	Out	Rem	n
BOD_5	309	21	87.9	97	166	19	147	83
COD	547	70	78.6	115	311	56	255	99
TSS	188	18	77.1	74	139	17	122	62
TP	10.6	4.6	48.3	94	5.7	3.5	2.2	81
TN	70	37.6	44.0	64	42.1	23.6	18.5	57
NH_4-N	56.4	10.6	78.9	94	27.8	7.1	20.7	85
NO_3-N	0.62	25.3		70	0.9	13.1		56
FC*	5.95	3.0	97.6	20				

VF constructed wetlands are primarily used to treat domestic or municipal sewage. However, in recent years VF systems have also been used to treat other types of wastewater (Table 4-18). VF constructed wetlands are also often used as part of hybrid systems treating various types of wastewater (see section 4.3).

Table 4-18. The use of VF systems for various types of wastewater other than sewage.

Waste type	Country	Reference
Perchlorate	USA	Tan et al. (2004)
Phenanthrene	Germany	Machate et al. (1997)
PAH	France	Cottin and Merlin (2006)
Herbicides	UK	McKinlay and Kasperek (1999)
Sulfides	Colombia	Giraldo and Zárate (2001)
Airport runoff	Canada	McGill et al. (2000)
Landfill leachate	Australia	Headley et al (2004)
Dairy	The Netherlands	Veenstra (1998)
Cheese dairy	Germany	Kern and Idler (1999)
Abattoir	Canada	AQUA TT
Mushroom farm leachate	Canada	AQUA TT
Composting leachate	Germany	Lindenblatt (2005)
Surface water	The Netherlands	Bloom and Verhoeven (2006)
Refinery wastewater	Pakistan	Aslam et al. (2007)
Aniline and N-compounds	Portugal	Novais and Martins-Dias (2003)

During the 1990s, the Tennessee Valley Authority developed so called "reciprocating" vertical flow constructed wetlands (e.g. Behrends, 2000;

Behrends et al., 1993, 1996, 2001). Reciprocating systems are usually designed as paired rectangular symmetrical cells, but can be designed as free-form asymmetrical cells that conform to the landscape. The process involves continuously pumping wastewater back and forth between adjacent cells such that each cell is partially-drained and refilled on a two hour cycle. This design provides controlled aeration of the substrate and exposes the substrate surface biofilms to atmospheric oxygen. The system has been successfully applied to municipal, industrial (explosives, food processing, airport de-icing water, acid mine drainage) as well as agricultural (aquaculture, swine feedlot) wastewaters (Behrends et al., 2001).

4.2.2.2 Upflow

In vertical-upflow constructed wetlands the wastewater is fed on the bottom of the filter bed. The water percolates upward and then it is collected either near the surface or on the surface of the wetland bed (Fig. 4-47). These systems have commonly been used in Brazil (Salati, 1987; Manfrinato et al., 1993; Salati et al., 1999) since the 1980s. The beds are filled with crushed rock on the bottom, the next layer is coarse gravel and the top layer is soil planted with Rice (*Oryza sativa*). This treatment system is called in Brazil "filtering soil" (Fig. 4-48). VF upflow systems were also studied under experimental conditions in Australia, New Zealand and Sweden (Breen, 1990; Rogers et al., 1990; Mitchell et al., 1990; Rowe, 1998; Farahbakhshazad and Morrison, 2003) or built in New Zealand (Rowe 1998). However, outside Brazil, the layer of soil is usually not used and beds are filled with gravel and usually planted with common species such as *Phragmites australis*.

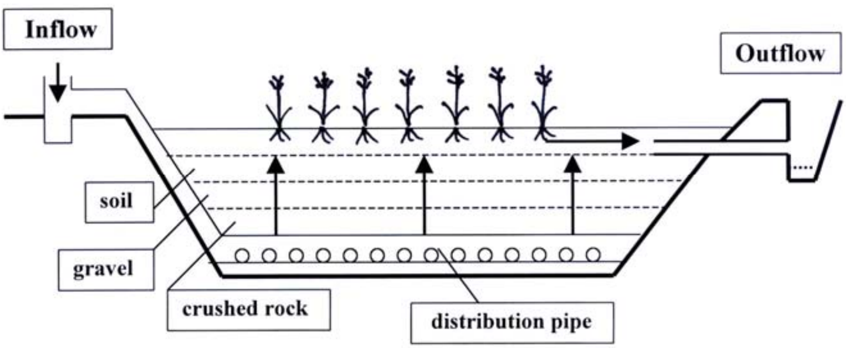

Figure 4-47. Schematic representation of a constructed wetland with vertical up-flow. From Vymazal (2001a) with permission from Backhuys Publishers.

Kassenga et al. (2003) reported the use of vertical upflow CWs to remove volatile organic compounds from contaminated water. The compounds tested were cis-1,2-dichloroethane and 1,1,1-trichloroethane. Amon et al. (2007) used upflow VF wetland to treat groundwater contaminated by chlorinated ethenes. Tan et al. (2004) studied the use of vertical upflow CW to treat perchlorate-contaminated water. Zhou et al. (2005) tested in China a combination of downflow and upflow constructed wetlands to remove phthalate acid esters, namely di-*n*-butyl phthalate (DBH). Gravel based beds were planted with *Canna indica* (Indian shot, downflow) and *Acorus calamus* (Sweet flag, upflow).

Figure 4-48. VF upflow constructed wetlands planted with rice (*Oryza sativa*) near Piracicaba, Brazil. Photos by Jan Vymazal.

4.2.2.3 Tidal flow

Tidal flow systems are a new form of VF system (Green et al., 1997; Revitt et al., 1997; Sun et al., 1999a,b; Zhao et al., 2004a; Cooper and

Cooper, 2005). Cooper (2005) pointed out that tidal flow systems were developed to try to overcome some of the problems that were seen in the early forms of VF systems related to clogging of the surface. Upflow systems have been used for about 20 years but they suffer from the problem that distribution is below the surface and hence hidden from the observers. In tidal flow systems at the start of the treatment cycle the wastewater is fed to the bottom of the bed into the aeration pipes. It then percolates upwards until the surface is flooded. When the surface is completely flooded the pump is then shut off, the wastewater is then held in the bed in contact with the microorganisms growing on the media. A set time later the wastewater is drained downwards and after the water has drained from the bed the treatment cycle is complete and air diffuses into the voids in the bed (Cooper, 2005).

4.3 Hybrid systems

Various types of constructed wetlands may be combined in order to achieve higher treatment effect, especially for nitrogen. There has been a growing demand in achieving fully-nitrified effluents but secondary treatment HF systems cannot do this because of their limited oxygen transfer capacity (Cooper et al., 1996; Vymazal et al., 1998a). VF systems have a much greater oxygen transport capacity and, therefore, provide much better conditions for nitrification. However, very limited or no denitrification occurs in VF systems (see Table 4-17). Therefore, there has been a growing interest in hybrid systems (also sometimes called combined systems). Hybrid systems used to comprise most frequently VF and HF systems arranged in a staged manner, however, all types of constructed wetlands could be combined (Table 4-19). In hybrid systems, the advantages of various systems can be combined to complement each other. It is possible to produce an effluent low in BOD, which is fully-nitrified and partly denitrified and hence has much lower total-N concentrations (Cooper 1999, 2001).

Many of these systems are derived from original hybrid systems developed by Seidel at the Max Planck Institute in Krefeld, Germany. The process is known as the Seidel system, the Krefeld system or the Max Planck Institute Process (MPIP) (Seidel 1965b, 1976, 1978). The design consists of two stages of several parallel VF beds ("filtration beds") followed by 2 or 3 HF beds ("elimination beds") in series. The VF stages were usually planted with *Phragmites australis*, whereas the HF stages contained a number of other emergent macrophytes, including *Iris, Schoenoplectus (Scirpus), Sparganium, Carex, Typha* and *Acorus*. The VF beds were loaded with pre-treated wastewater for 1 to 2 days, and were then allowed to dry out for 4 to 8 days. The thin crust of solids that forms on top of the VF beds is mineralized during the rest period and achieves an equilibrium thickness.

In the early 1980s, several hybrid systems of Seidel's type were built in France with the systems at Saint Bohaire and Frolois being the best described (Boutin, 1987; Vuillot and Boutin, 1987; Liénard et al., 1990, 1998). A similar system was built in 1987 in the United Kingdom at Oaklands Park (Burka and Lawrence, 1990, Fig. 4-42, Table 4-19). In the 1990s and the early 2000s, VF-HF systems were built in many countries, e.g., Slovenia (Urbanc-Berčič and Bulc, 1994), Norway (Mæhlum and Stålnacke, 1999), Austria (Mitterer-Reichmann, 2002), Ireland

Table 4-19. Examples of hybrid constructed wetlands used for various types of wastewater.

Type of CW	**Country**	**Type of (waste)water**	**Reference**
VF-HF	France	Milking parlor	Liénard et al. (1998)
VF-HF	UK	Sewage	Burka and Lawrence (1990), Cooper (2001)
VF-HF	Slovenia	Sewage	Urbanc-Berčič and Bulc (1994), Urbanc-Berčič et al. (1998)
VF-HF	USA	Sewage	House and Broome (2000)
VF-HF	Slovenia	Landfill leachate	Bulc (2006)
VF-HF	France	Cheese dairy	Reeb and Werckmann (2005)
VF-HF	France	Compost leachate	Reeb and Werckmann (2005)
VF-HF	Thailand	Pig farm wastewater	Kantawanichkul and Neamkam (2003)
VF-HF	Estonia	Sewage	Öövel et al. (2007)
VF-HF	Belgium	Sewage	Lesage (2006)
VF-HF	Poland	Slaughterhouse	Soroko (2005)
VF-HF	Japan	Milking parlor	Kato et al. (2006)
HF-VF	Denmark	Sewage	Brix et al. (2003)
HF-VF	Poland	Sewage	Ciupa (1996), Obarska-Pempkowiak et al. (2005)
HF-VF	S. Africa	Sewage	Wood and Hensman (1989)
HF-VF	Nepal	Hospital	Laber et al. (1999)
HF-VF	Mexico	Sewage	Belmont et al. (2004)
FWS-HF	Taiwan	Polluted river	Jing et al. (2001)
FWS-HF	Taiwan	Shrimp aquaculture	Lin et al. (2003, 2005)
FWS-HF	Taiwan	Fish aquaculture	Lin et al. (2002)
FWS-HF	China	Industrial	Wang et al. (1994)
HF-FWS	Canada	Landfill leachate	Kinsley et al. (2006)
HF-FWS	Italy	Winery wastewater	Masi et al. (2002)
HF-FWS	Norway	Landfill leachate	Mæhlum et al. (1999)
HF-FWS	Canada	Sewage	Laouali et al. (1996)
HF-FWS	Kenya	Sewage	Nyakang'o and van Bruggen (1999)
VF-VFD	Brazil	Sewage	Nogueira et al. (2000)
HF-VF-HF	Poland	Sewage	Tuszyńska and Obarska-Pempkowiak (2006)
VF-HF-FWS-P	Italy	Winery wastewater	Masi et al. (2002)
VF-HF-FWS-P	Estonia	Sewage	Mander et al. (2003)
HF-VF-HF-FWS	Italy	Sewage	Pucci et al. (2004)

(O'Hogain, 2003), Belgium (Rousseau, 2005), Estonia (Öövel et al., 2007), USA (House and Broome, 2000), Thailand (Kantawanichkul and Neamkam, 2003) (see also Table 4-19).

In Tables 4-20 and 4-21, performance examples of VF-HF hybrid systems are presented. The results indicate very good removal for organics (BOD_5 and COD) and TSS while removal of nitrogen is enhanced with no nitrate increase at the outflow. Data from Oaklands Park (Table 4-20) indicate that nitrification takes place in VF beds while denitrification in HF beds. However, the nitrification was not complete at the end of the 2nd VF stage indicating that the VF beds were a little too small. The same is seen for De Pinte-Zevergem system (Table 4-21, Fig. 4-49).

Most VF-HF constructed wetlands have been used to treat municipal sewage but there are also examples of treatment of other types of wastewater (Table 4-19, Fig. 4-50).

Table 4-20. Treatment performance of a hybrid system at Oakland Parks (see also Fig. 4-42). Concentrations in mg L^{-1}. From Cooper (2001) with permission from Backhuys Publishers.

	Inflow	**1st VF Out**	**2nd VF Out**	**1st HF Out**	**2nd HF Out**
BOD_5	285	57	14	15	7
TSS	169	53	17	11	9
NH_4^+-N	50.5	29.2	14.0	15.4	11.1
NO_x^-N	1.7	10.2	22.5	10.0	7.2
Ortho-P	22.7	18.3	16.9	14.3	11.9

Figure 4-49. VF-HF constructed wetland (2,250 m^2) at De Pinte-Zevergem, Belgium planted with *Phragmites australis*. Photo by Lenka Kröpfelová.

Table 4-21. Treatment performance of VF-HF constructed wetland at De Pinte-Zevergem, Belgium. Concentrations in mg l^{-1}. Data from VMM (2006).

	Inflow				Outflow			
	2002	**2003**	**2004**	**2005**	**2002**	**2003**	**2004**	**2005**
BOD_5	34	81	91	83	3.0	8.0	6.0	6.0
COD	113	264	258	250	36	49	45	42
TSS	33	60	80	37	4.0	8.0	4.0	4.0
TP	2.8	6.1	6.0	6.2	1.1	3.6	3.6	3.3
TN	23.2	59.1	57.2		12.9	37	31.8	
TKN	21.8	58.9	56.6		11.7	34.1	29.1	
NO_x-N	1.4	0.2	6.0		1.2	2.9	2.7	

Figure 4-50. VF-HF hybrid constructed wetlands for composting plant leachate treatment in Bretagne, France. The 1st VF stage (4 x 120 m^2) is planted with *Phragmites australis*, 2nd VF stage (4 x 72 m^2) is planted with *Typha latifolia*, *P. australis*, *Scirpus lacustris* and *Phalaris arundinacea* in each bed and 3rd HF stage (2 x 290 m^2) is planted with *Carex acuta*, *Iris pseudacorus*, *Acorus calamus*, *S. lacustris*, *Juncus effusus*, *Mentha aquatica* and *Sparganium erectum*. From Reeb and Werckmann (2005) with permission from Backhuys Publishers.

In the mid-1990s, Johansen and Brix (1996) introduced a HF-VF hybrid system with a large HF bed placed first and a small VF bed as the second stage (Fig. 4-51). In this system nitrification takes place in the vertical-flow stage at the end of the process sequence. If nitrate removal is needed it is then necessary to pump the effluent back to the front end of the system where denitrification can take place in the less aerobic horizontal-flow bed using the raw feed as a source of carbon needed for denitrification (Cooper et al., 1996). Brix et al. (2003) pointed out that special care must be taken not to affect the performance of the sedimentation tank or the nitrifying capacity of the VF bed by recycling too large volume of wastewater. Results from the first HF-VF system in Denmark at Bjødstrup-Landborup are presented in Table 4-22.

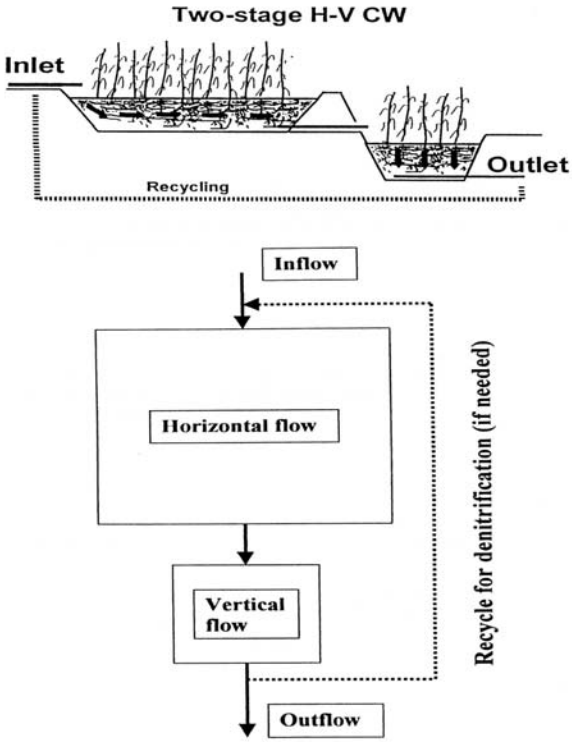

Figure 4-51. Schematic arrangement of the HF-VF hybrid system according to Brix and Johansen. From Vymazal (2001a) with permission from Backhuys Publishers.

Similar system (448 m^2 HSF followed by 44 m^2 VF) was reported, for example, from Sobiechy in Poland (Ciupa, 1996). Also Obarska-Pempkowiak et al. (2005) reported on HF (1,610 m^2) – VF (520 m^2) hybrid system in Sarbsk, Poland. In fact, one of the first HF-VF systems was reported from South Africa by Wood and Hensman (1989). The system was built at Grootvlei power station in Transvaal to treat bio-filter plant effluent. At the average HLR of 7.5 cm d^{-1} the removal amounted to 79% for TN, 86% for PO_4-P and 67% for COD with an outflow TN, phosphate and COD concentrations of 1.9 mg l^{-1}, 0.8 mg l^{-1} PO_4-P and 10 mg l^{-1} COD.

Table 4-22. Performance of a HF (456 m^2) – VF (30 m^2) constructed wetland (55 PE) at Bjødstrup-Landborup, Denmark. From Brix et al. (2003) with permission from Backhuys Publisher.

	Period 1997-2000		**June 2000, with nitrified effluent recycling**		
	Inflow	**Outflow**	**HF Inflow**	**HF Outflow**	**VF Outflow**
COD	496	70	423	54	41
BOD_5	267	4.3			
TP	13.4	0.8			
TN	79	43	58	25.4	17.7
NH_4-N		11.2	18	14.7	2.5
NO_3-N			1.3	2.5	9.8

The importance of recirculation in HF-VF systems could be seen from the results of the system at Dhulikhel, Nepal (Fig. 4-52, Table 4-24) where VF bed provides full nitrification but no denitrification takes place in this bed.

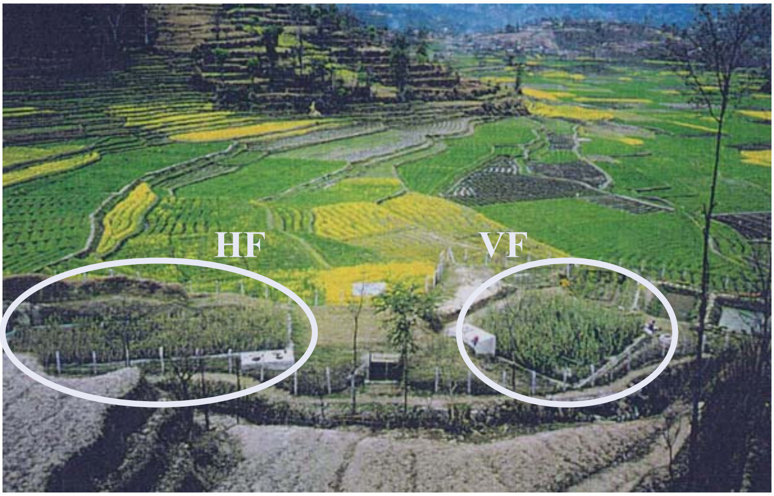

Figure 4-52. VF-HF constructed wetland at Dhulikhel, Nepal planted with *Phragmites karka*. From Shrestha (1999), with permission from Universität für Bodenkultur, Vienna, Austria.

Table 4-23. Performance of the HF (140 m^2) – VF (120 m^2) hybrid system at Dhulikhel, Nepal (modified from Laber et al., 2003). ST=septic tank, values in mg l^{-1}, *E. coli* in log_{10}CFU 100 ml^{-1}. Eff = treatment efficiency of individual stages (%).

	STin	STout	Eff	HFout	Eff	VFout	Eff
BOD_5	118	67	43	25	63	2	92
COD	261	162	38	45	72	10	78
TSS	159	57	64	19	67	1.5	92
NH_4-N	32	32	0	27	16	0.1	96
NO_3-N	0.2	0.2	0	0.4		27	
TP	4.6	4.4	4	2.6	41	1.4	46
E. coli	7.2	6.2	1.0	3.6	2.6	1.3	2.3

During the 1990s, various types of constructed wetlands other than HF and VF started to be combined in order to achieve the best treatment effect possible (for examples see Table 4-19). In Figures 4-53 and 4-54, FWS-HF constructed wetlands are presented. The Yantian system, despite high HLR of 36 cm d^{-1} for HF part, provided a good quality effluent, particularly for TSS (99% removal) and COD (81%) (Wang et al., 1994). The system at Otorohanga in New Zealand (Fig. 4-54) provides tertiary treatment after a stabilization pond at HLR of 5 cm d^{-1}.

Figure 4-53. FWS-HF hybrid constructed wetlands at Yantian Industry Area in Baoan District, Shengzhen City. The system consists of up flow pond (247 m^2), three FWS cells (275 m^2 each) with *Eichhornia crassipes* and two HF beds (805 m^2 each) planted with *Phragmites australis*. Photo by Jan Vymazal.

Figure 4-54. A 3 ha tertiary FWS-HF constructed wetland at Otorohanga, New Zealand. FWS part in front is planted with *Eleocharis sphacelata* and *Scirpus tubernaemontani*, HF part in the back is planted with *Baumea articulata* and *Carex secta*. Photo by Jan Vymazal.

Masi et al. (2002) reported the use of HF-FWS constructed wetland to treat winery wastewaters with high organic load (Fig. 4-55, Table 4-24). The inflow organic loading is extremely high but the system´s removal efficiency is excellent. The FWS part as a final stage of the treatment process usually results in increase in TSS concentration, especially during the growing season. Increased TSS concentrations at the outflow are a consequence of the growth of planktonic algae.

Figure 4-55. HF (450 m^2) - FWS (850 m^2) constructed wetland at Casa Vincicola, Cecchi, Italy. HF part is planted with *Phragmites australis*, FWS part is planted with dominance of *Nymphae* sp., *Typha* sp. and *Juncus* sp. Photos by Fabio Masi, with permission of IRIDRA S.r.l.

Table 4-24. Casa Vinicicola Cecchi, Italy – treatment performance for the period 13.2. 2001 – 11.3. 2003 (based on Masi et al., 2002).

	Concentration (mg l^{-1})					Loading (kg ha^{-1} d^{-1})				
	BOD_5	COD	TSS	TN	TP	BOD_5	COD	TSS	TN	TP
Inflow	1833	3906	213	18.9	4.7	1336	2847	155	13.8	3.4
HF out	49.4	131	13.3	4.8	1.5	36	96	9.7	3.5	1.1
FWS out	25.4	84	23.4	3.5	1.3	18.5	61.2	17.1	2.6	0.95

Laouali et al. (1996) reported the use of combination of HF and FWS wetlands in Montreal, Canada (Fig. 4-56). The system has been designed as three-stage wetland; it consists of: 1) two parallel HF beds (stage I) planted with *Phragmites australis* (200 m^2 each), 2) 300 m^2 pond (stage II) divided into three 100 m^2 sections in series planted with *Scirpus lacustris*, *Typha latifolia* and *Iris versicolor*, and 3) 100 m^2 pond (stage III) divided into two 50 m^2 sections in series planted with *Mentha aquatica* and *Elodea canadensis*. As for most systems where the FWS stage is at the end of the treatment line, removal of nutrients and bacteria improves at this stage but suspended solids and organics exhibit a slight increase due to algal growth (Table 4-25).

Figure 4-56. HF-FWS constructed wetland in Biosphère de Montréal, Canada. Left: HF stage, Right: FWS stage. Photos by Jan Vymazal.

Table 4-25. Treatment performance (values in mg l^{-1}) of a hybrid system in Montréal (calculated from Laouali et al., 1996).

	BOD_5	COD	TSS	TKN	NH_4-N	TP
Inflow	130	283	67	52.8	37.3	7.1
HF out	16.2	51	19.8	27.7	19.8	0.73
FWS I out	3.0	39	22.5	16.7	10.7	0.30
FWS II out	5.5	39	18.5	10.2	6.1	0.17

Recently, even more complex hybrid constructed wetlands have been used. The system in Daržlubie (Fig. 4-57) in Poland consists of a combination of HF bed (1200 m^2), cascade of 5 alternate HF and VF beds (total are of 270 m^2) and HF (II) bed (500 m^2). After this point 50% of the flow is directed to two VF(II) beds (total area 500 m^2) and the final stage of the treatment system is 1000 m^2 HF bed where outflow from VF(II) and HF(II) are combined (Obarska-Pempkowiak, 1999). The system was designed to serve 750 inhabitants and all beds are planted with *Phragmites australis*.

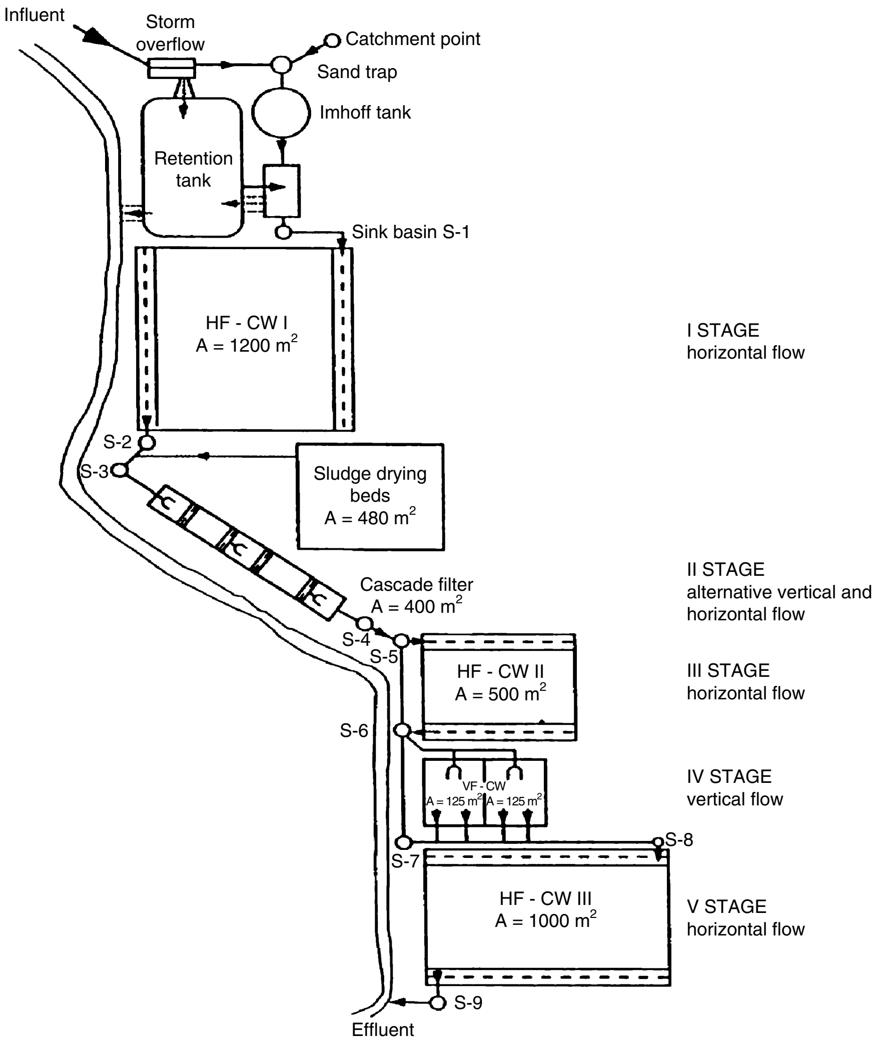

Figure 4-57. Hybrid constructed wetlands at Darzlubie, Poland. From Obarska-Pempkowiak (1999) with permission from Backhuys Publishers.

Masi et al. (2002) reported the use of VF-HF-FWS-pond hybrid system to treat winery wastewaters in Ornellaia, Italy (Table 4-26). The system consists of two VF (90 m^2 each), 102 m^2 HF bed, 148 m^2 FWS and 338 m^2 pond. Mander et al. (2003) reported the use of a VF-HF-FWS-pond system for municipal sewage at Kõo, Estonia (Fig. 4-58).

Table 4-26. Performance of hybrid treatment system at Ornellaia treating winery wastewater. Concentrations in mg l^{-1}. Based on Masi et al. (2002).

	Inflow	VFout	HFout	FWS out
BOD_5	425	337	286	28.6
COD	1003	690	431	79
TSS	103	42	24	47.5
NH_4-N	26.6	8.7	4.7	2.7

The Belgian researchers has developed and tested an extensive MHEA® (Mosaïques Hiérarchisées d'Ecosystémes Artificiels) system consisting of a series of FWS and sub-surface flow units planted often with *Typha* spp. and woody species such as *Salix viminalis, Populus alba* or *Alnus glutinosa* (Radoux, 1982, 1994; Radoux et al., 2003; Cadelli et al., 2005).

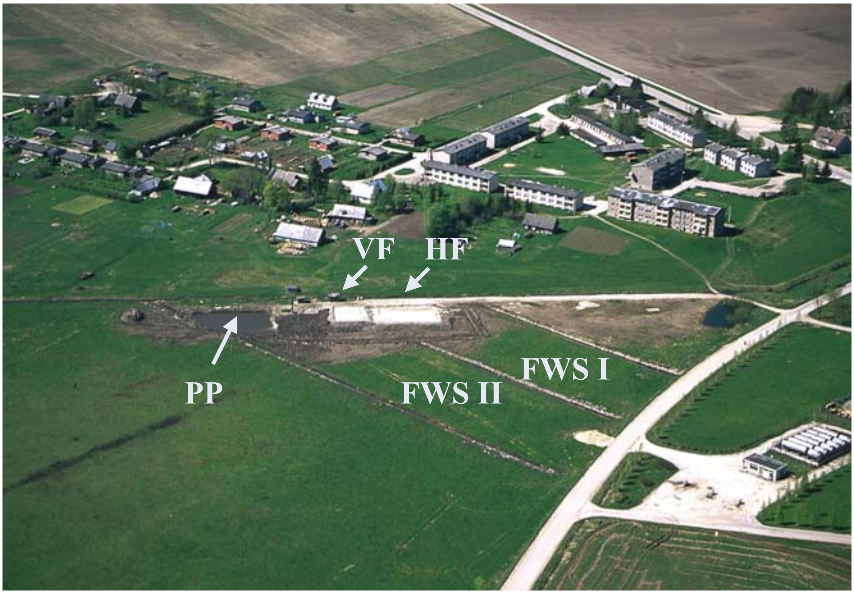

Figure 4-58. Aerial picture of VF (2 x 64 m^2) - HF (365 m^2) - FWS (3600 and 5000 m^2) – polishing pond (PP, 500 m^2) system at Kõo, Estonia. VF and HF parts are planted with *Phragmites australis*, FWS parts are planted with *Typha latifolia*. Photo by Ülo Mander, with permission.

4.4. Zero discharge systems

A novel CW system based on willows has been developed as a sewage disposal solution in rural areas in Denmark (Gregersen and Brix, 2001; Brix and Gregersen, 2002). Main attributes of the willow wastewater cleaning facilities are that the systems have zero discharge of water (because of evapotranspiration) and part of the nutrients can be recycled via the willow biomass. Furthermore, the harvested biomass may be used as a source of bioenergy (Brix and Arias, 2005). The willow wastewater cleaning facilities generally consist of about 1.5 m deep high-density polyethylene-lined basins filled with soil and planted with clones of willow (*Salix viminalis* L.) (Figs. 4-59 and 4-60). The surface area of the systems depends on the amount and quality of the sewage to be treated and the local annual rainfall. For a single household in Denmark the area needed typically is between 120 and 300 m^2. Settled sewage is dispersed underground into the bed under pressure. The stems of the willows are harvested on a regular basis to stimulate the growth of the willows and to remove some nutrients and heavy metals.

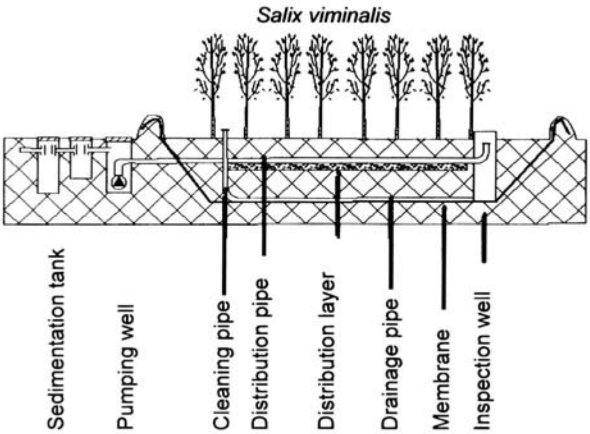

Figure 4-59. Sketch of a willow system with no outflow (evaporation system) (Brix and Arias, 2005). Reprinted from *Water Science and Technology*, volume 51, issue 9, pp. 1-9, with permission of the copyright holder, IWA.

Brix and Arias (2005) pointed out that removal of water from the systems occurs by evaporation from the soil and plant surface, and transpiration. The following factors are important for maximising evaporative loss of water: High energy input (solar radiation), high air temperatures, low relative humidity in the air, exchange of air (wind), canopy resistance, stomata resistance, and leaf area index. Factors like the 'oasis' effect, which is the phenomenon where warmer and dry air in equilibrium with dry areas flows across a vegetation of plants with a high water availability (Rosenberg, 1969). The vegetation experiences enhanced evaporation using sensible heat from the air as well as radiant energy, and air is cooled by this process. In addition, the so-called 'clothesline' effect, where the vegetation height is greater than that of the surroundings (different roughness conditions), may increase evaporative water loss (Allen et al., 1998). This occurs where turbulent transport of sensible heat into the canopy and transport of vapor away from the canopy is increased by the 'broadsiding' of wind horizontally into the taller vegetation. In addition, the internal boundary layer above the vegetation may not be in equilibrium with the new surface. Therefore, evapotranspiration from the isolated expanses, on a per unit area basis, may be significantly greater than the calculated potential evapotranspiration. Examples of the clothesline or oasis effects would be evapotranspiration from a single row of trees surrounded by short vegetation or surrounded by a dry non-cropped field, or evapotranspiration from a narrow strip of cattails (a hydrophytic vegetation) along a stream channel.

The main characteristics of the willow systems are (Ministry of Environment and Energy 2003a):

- For a single household (5 PE) system, the sewage has to be pre-treated in a 2- or 3-chamber sedimentation tank with a minimum volume of 2 m^3 before discharge into the willow system.
- Closed willow systems are generally constructed with a width of 8 m, a depth of minimum 1.5 m, and with 45 degree slopes on the sides.
- The total annual water loss from the systems is assumed to be 2.5 times the potential evapotranspiration at the location as determined by climatic parameters.
- The necessary area of the systems is determined by the amount of wastewater, the 'normal' precipitation, and the potential evapotranspiration at the location of the system.
- The bed is enclosed by a water tight membrane and wastewater is distributed underground within the system by a level controlled pump.
- A drainage pipe is placed in the bottom of the bed. The pipe can be used to empty water from the bed if salt accumulates after some years.
- One third of the willows are harvested every year to keep the willows in a young and healthy state with high transpiration rates.

Figure 4-60. Willow based constructed wetland at Vråvej, Denmark. One third of aboveground biomass is harvested every year. Photos by Jan Vymazal.

Chapter 5

HORIZONTAL FLOW CONSTRUCTED WETLANDS

5.1 Technology development

The technology of wastewater treatment by means of constructed wetlands with horizontal sub-surface flow was started in Germany based on research by Käthe Seidel commencing in the 1960s (e.g., Seidel, 1961, 1964, 1965 a,b, 1966) and by Reinhold Kickuth in the 1970s (e.g., Kickuth, 1977, 1978, 1981).

Seidel intensified her trials to grow helophytes and hydrophytes in wastewater and sludge of different origin and she tried to improve the performance of rural and decentralized wastewater treatment which was either septic tank or pond system with low purification effect. She planted helophytes into the shallow embankment of tray-like ditches and created artificial trays and ditches grown with helophytes. The system worked effectively and Seidel named this early system the "hydrobotanical method" (Börner et al., 1998). She improved further this method by using sandy soils with high hydraulic conductivity in sealed module type basins planted with different species of helophytes. To overcome the anaerobic septic tank system she integrated a stage of primary sludge filtration in vertically percolated sandy soils grown with *Phragmites australis* (Seidel, 1965b). The first vertical stage was so called "filter or filtration bed". The second horizontal flow stage was called "elimination bed" (Fig. 5-1) and was usually planted with *Scirpus lacustris*. However, too much significance was attracted to the absorption of nutrients by plants and this has been a target for criticism (e.g., Nümann, er, 1970). However, numerous variants based on this fundamental concept have been developed over the years. The system itself was revived later and now it is called a “hybrid system”. Both vertical and horizontal flow stages have been used separately as well.

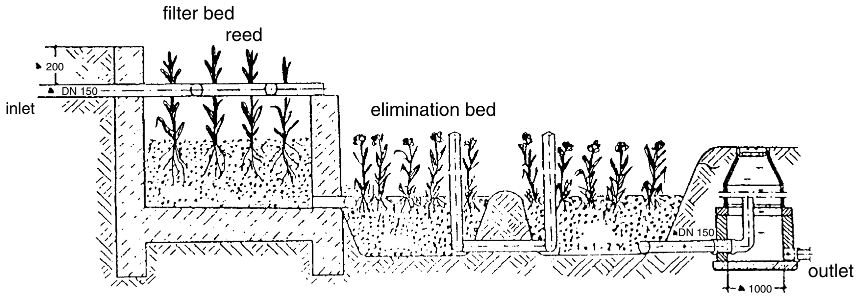

Figure 5-1. Schematic representation of the constructed wetland system designed by Seidel. From Börner et al. (1998) with permission from Backhuys Publishers.

In the constant search for analytical support from universities Seidel came to know the soil scientist R. Kickuth from Göttingen University in the mid-1960s. A fruitful collaboration developed (Kickuth, 1969) but it was interrupted after a few years due to personal reasons (Wissing, 1995). Kickuth´s concept based on the use of cohesive soils with high clay content - “Wurzelraumentsorgung”, “Root Zone Disposal System” or “Root Zone Method”- was closer to the traditional understanding of a land treatment of sewage. However, his statement that hydraulic permeability of soils will increase by several orders of magnitude by root and rhizome growth and movement of *Phragmites australis* stems turned out to wrong in the field (Bucksteeg, 1986). In addition, hydraulic problems and the resulting operational difficulties with the roots’ growth into the substrate proved to be substantial (Ministerium für Umwelt, 1990). This gave a harsh setback to the scientific and official acknowledgement of constructed wetlands for wastewater treatment systems (Börner et al., 1998).

The first full-scale HF Kickuth’s type CW for treatment of municipal sewage was put in operation in 1974 in the community Liebenburg-Othfresen (Kickuth, 1977). The area of about 22 ha was originally used to dump waste material (silt, clay and dross) derived from mining of iron ore. It contained settlement ponds for separation of silt and clay which were filled with clay, gravel, and chalk when mining ceased in 1962. In 1969, the local authority proposed to use part of the area for sewage treatment. The anaerobic maturation ponds were rejected and Kickuth recommended the so called Root-Zone Method to be constructed on higher ground of the area

(Boon, 1986). The results from this system were exceptionally good but the validity of these data has later been seriously questioned (Brix, 1987b).

In 1981, HF constructed wetland was built in Weinitzen near Graz, Austria and this system has been monitored until 1986 (Tischler et al., 1985). In 1983, a large experimental 4-bed HF system was built in Mannersdorf Austria (see also Fig. 7-1). Since June 1984, bed 1 was fed with raw sewage, bed 2 with settled sewage and beds 3 and 4 with biologically-treated sewage effluent. The research unit has been intensively investigated between 1983 and 1989 (Haberl and Perfler, 1989, 1990).

In 1983, German ideas were introduced in Denmark and by 1987 about 80 HF CWs were built (Fig. 5-2, Brix, 1987b, 1998, 2003a). Despite problems with surface-flow soil-based systems exhibited high treatment effect for organics and suspended solids if reed bed area 3-5 m^2 PE^{-1} was used.

Figure 5-2. Two of the first HF constructed wetlands built in Denmark in 1984: top: Ferring (2,000 m^2, 500 PE), bottom: Knudby (418 m^2, 65 PE). Photos by Hans Brix, with permission.

In order to overcome the overland flow Danish systems were designed with a low aspect ratio (length/width ratio) and thus have very long inlet trenches and a comparably short passage length. However, the design with a very long inlet trenches caused problems with water distribution and, therefore, the inlet trench was subdivided into two or more separate units that could be loaded separately in order to get better control on the distribution of water (Brix, 1998).

In 1985, following visits to existing German and Danish systems, first two HF CWs were built in the United Kingdom (here called Reed Bed Treatment Systems, Fig. 5-3) and by the end of 1986 more than 20 HF CWs were designed (Cooper and Boon, 1987). The major change in the design was the use of very coarse filtration material (5-10 mm fraction) which ensured sub-surface flow similarly to former design by Seidel (1965b). Also, specific area of 5 m^2 PE^{-1} was used in the United Kingdom. During the 1990s, HF CWs were built in many European countries and HF CWs became the most widely used concept in Europe in the 1990s (Vymazal et al., 1998a).

Figure 5-3. HF constructed wetland at Acle, United Kingdom, one of the two CWs built in 1985. Photo by Jan Vymazal.

In North America, the first mesocosm and pilot scale constructed wetlands with horizontal sub-surface flow were built in Seymour, Wisconsin to treat municipal wastewater, in 1972 and 1974 (Spangler et al., 1976; Fetter et al., 1976). In 1973, another pilot constructed wetland system was constructed at Brookhaven National Laboratory, New York. This pilot treatment system combined marsh wetlands with a pond and a meadow in series and was designated as the marsh/pond/meadow (MPM) treatment system (Small and Wurm, 1977). In the early 1980s, Wolverton and his co-workers at the NASA, who had been researching floating plant treatment systems (e.g. Wolverton and McDonald, 1976), began to focus on subsurface constructed wetlands (there called “rock reed filters”) (e.g., Wolverton, 1982, 1987; Wolverton et al., 1984a). Gersberg and co-workers started to study horizontal subsurface constructed wetlands in California (Gersberg et al., 1983, 1984) and in 1986, Tennessee Valley Authority started to evaluate this system as well (Steiner et al., 1987, see also Section 7.2.3). North America was slower to adopt subsurface technology as compared to Europe. However, in recent years the use of these systems has drawn more attention and it is estimated that there are about 8,000 subsurface constructed wetlands at present (Kadlec, 2003). However, the information on HF CWs in North America as compared to FWS systems is very limited in the literature. The subsurface technology has also been adopted in Canada (Pries, 1994) and Mexico (e.g., Durán de Bazúa et al., 2000).

The potential use of aquatic and wetland macrophytes for wastewater treatment was evaluated in Australia by Mitchell during the mid 1970s (Mitchell, 1976). In 1980, the assimilative capacity of wetlands for sewage effluent was evaluated (Bavor et al., 1981) and Finlayson and co-workers performed pilot-scale experiments on the use of sub-surface constructed wetlands for the treatment of piggery wastes and abattoir wastewater (Finlayson and Chick, 1983; Finlayson et al., 1987). Extensive pilot-scale experiments were also carried out at University of Western Sydney (Bavor et al., 1987, Fig. 5-4).

However, despite several pilot projects in Australia in the 1980s, this wastewater-treatment technology has not widely been adopted in Australia (Greenway and Simpson, 1996; Greenway and Woolley, 1999, 2001; QDNR, 2000). Despite free water surface constructed wetlands being more common in Australia, HF systems have also been popular in some rural parts of the country for in-site treatment of domestic sewage (Davison et al., 2006). According to the survey carried out in 1999 in New Zealand, constructed wetlands had been adopted enthusiastically by many New Zealand communities as a cost-effective means of secondary and tertiary wastewater treatment (Tanner et al., 2000). The survey revealed that there were about 80 constructed wetlands for wastewater treatment excluding those treating stormwaters and farm dairy wastes. Surface flow CWs were most common (45%) followed by subsurface flow and hybrid systems (35% and 14%, respectively).

Figure 5-4. Research HF constructed wetlands located at the University of Western Sydney, Hawkesbury, NSW, Australia. Photo by Jan Vymazal.

Since the mid 1980s, the concept of using constructed wetlands has gained increasing support in South Africa. By 1990, there were approximately 30 systems either in operation or under construction. These have been designed to serve a number of functions from treating raw sewage and secondary domestic effluents, upgrading septic tank and oxidation pond effluents, storm waters, agricultural and aquaculture wastes and a variety of industrial and mining wastewaters. Most of these systems were designed as HF constructed wetlands (Wood, 1990; Wood and Hensman, 1989; Batchelor et al., 1990). However, after the mid 1990s, the information from the South Africa diminished so it is not possible to find out if HF constructed wetlands became more widely spread there. On the other hand, at the end of the 20th century constructed wetland became more popular in tropical parts of Africa and there are now many fine examples of all types of constructed wetlands including HF systems treating municipal sewage as well as industrial wastewaters and mine drainage waters in (e.g., Byekwaso et al., 2002; Kaseva, 2004, Fig. 5-5).

The traditional expertise of Asian farmers in recycling human and animal wastes through aquaculture and the practices intuitively developed by them for recovering nutrients from wastes by aquatic macrophytes (mostly Water hyacinth) propagated over waste-fed ponds gave a good basis for more engineered systems (Abassi, 1987). As early as in 1969, Sinha and Sinha reported on the use Water hyacinth to treat digested sugar factory wastes. During the 1970s and 1980s numerous experiments with Water hyacinth were conducted across Asia to treat various types of wastewater, e.g. from dairies, palm oil production, distillery, natural rubber production, tannery,

Figure 5-5. Constructed wetland at Kasese, planted with *Phragmites mauritianus* in Uganda treating effluent from cobalt recovery processing plant. Photo by Frank Kansiime, with permission.

textile, electroplating, pulp and paper production, pesticide production and heavy metals (Abassi, 1987). However, the first information about the use of constructed wetlands with horizontal subsurface flow appeared in the literature only in the early 1990s (Juwarkar et al., 1994). During the IWA conference in China in 1994, several papers on horizontal CWs from Asia, and especially China, were presented (e.g., Yang. et al., 1994; Wang et al., 1994; Xianfa and Chuncai, 1994) and, therefore, it is probably a lack of literature information which made the Asian systems "unrecognized". At present, HF CWs are in operation, among others, in India, China, Korea, Taiwan, Japan, Nepal, Malaysia or Thailand for various types of wastewater (e.g., Billore et al., 1999; Lee et al., 2004; Lin et al., 2005; Ham et al., 2004; Bista et al., 2004; Kantawanichkul and Somprasert, 2005).

The information on the use of constructed wetlands with horizontal flow in South and Central America is limited but these systems are apparently in operation Brazil, Chile, Colombia, Ecuador, Uruguay, (e.g.; Mariangel and Vidal, 2007; Perdomo, pers. comm., Platzer et al., 2002; Philippi et al., 2006, Fig. 5-6).

Figure 5-6. Newly-planted HF constructed wetland San Jacinto, Uruguay, treating wastewater from a meat processing facility shortly after plantation of *Typha domingensis* in 2006. Total area of four beds is 1.5 ha, average flow 3000 m^3 d^{-1}. Photo by Silvana Perdomo, with permission.

5.2 Major design parameters

5.2.1 Pretreatment

If HF constructed wetlands are used as a secondary treatment stage, pretreatment is essential in order to achieve good treatment performance of the system. In its most general form, pretreatment can be thought of as a processing sequence by which the distribution of the size particles found in wastewater is altered. The principal pretreatment operations and processes used for constructed wetlands are identified in Table 5-1. The following section deals with pretreatment units commonly used for domestic/municipal wastewaters. Industrial and agricultural wastewaters may require more complex pretreatment (e.g., see Fig. 7-47).

The stage of pretreatment where large debris is removed from wastewater is often called preliminary treatment. The most common preliminary treatment units are screens and grit removal chambers. Screening is typically the first unit operation encountered in a wastewater treatment plant. Screens are designed to intercept and retain large pieces carried on by wastewater, such as branches, leaves, fabrics, plastics as well as coarse solids such as cigarette butts, corks, pieces of vegetable and fruits or paper. Screens could be coarse, fine or micro depending on the clear openings between bars.

Large treatment plants commonly use both coarse and fine screens in succession but small treatment systems, such as constructed wetlands, mostly use coarse screens only.

Table 5-1. Typical operations and processes used for the pretreatment of domestic and municipal wastewater during the wastewater treatment in constructed wetlands with horizontal sub-surface flow, and the size particles affected. Modified from Tchobanoglous (2003).

Operation/process	Application/occurrence	Particle size affected
Screening, coarse	Used to remove large particles such as sticks, rags, and other large debris from untreated wastewater by interception	> 15 mm
Screening, fine	Removal of small particles	1.5 – 6.0 mm
Screening, micro	Removal of small particles	> 0.025 mm
Comminution	Used to cut up or grind large particles remaining after coarse screening into smaller particles of a more uniform size	6 mm
Gravity separation	Removal of settleable solids and floating material	> 0.040 mm
Grit removal	Removal of grit, sand, and gravel	> 0.15 mm
Oil and grease removal	Removal of oil and grease from individual discharges	
Imhoff tank	Used for the removal of suspended materials from household wastewater by sedimentation and flotation	> 0.040 mm
Septic tank	Used for the removal of suspended materials from household wastewater by sedimentation and flotation	> 0.040 mm

The screening element typically consists of parallel bars or rods or wires of uniform size. A screen composed of parallel bars or rods is called a bar screen (rack). Typically, bar racks in coarse screens have clear openings (space between bars) of 25-50 mm and they are fixed under 30-60^o angle in the inflow trench (Fig. 5-7). The screenings removed by hand raking are usually placed on a perforated plate to dewater (Fig. 5-8). For small treatment systems, manually-cleaned bar screens are typically used. However, mechanically-cleaned screens are used as well, especially for larger systems (Fig. 5-9). Screenings are usually composed of fabrics (50%), paper (20-30%), plastics (5-10%), rubber and prophylactics (2%), vegetable and fruit pieces (2-3%) and undissolved feces (2-3%) (Dohánoys et al., 2004). Care must be taken when disposing the screenings because the retained matter includes pathogenic fecal material.

Figure 5-7. Bar screens. Constructed wetland Čejkovice, Czech Republic. Photo by Lenka Kröpfelová.

Figure 5-8. Manually-cleaned bar screens with dewatering plate on top of the screen, left: CW Slavošovice, right: CW Malhotice, Czech Republic. Photos by Lenka Kröpfelová.

Figure 5-9. Mechanically-cleaned screens at constructed wetland Beja, Portugal. Photo by Lenka Kröpfelová.

Grit is composed of sand, gravel, cinders, or other heavy solid materials that have subsiding velocities or specific gravities substantially greater than those of the organic putrescible solids in wastewater (Crites and Tchobanoglous, 1998). Grit chambers (or grit trap) are predominantly used for combined sewer systems where stormwater runoff carries large amounts of grit. Under ideal conditions, grit chambers should retain only inorganic particles larger than about 0.2 mm without organic material. Typically, grit chambers are located after the bar screens but in some treatment plants, grit chambers precede the screening facility (Fig. 5-10). There are three types of grit chambers: horizontal flow, vertical and aerated. However, small system nearly always use horizontal flow grit chambers.

The horizontal flow channel-type grit chamber is the oldest and the most simple unit. The flow velocity in horizontal grit chambers should be maintained between 0.15 and 0.45 m s^{-1} with optimal velocity of 0.3 m s^{-1} providing sufficient time for grit particles to settle to the bottom of the channel. The velocity of flow is controlled by the dimensions of the unit and the use of special weir sections at the effluent end.

The chamber grit removal unit (Fig. 5-11) is one of the commonly used options. It consists of two or more narrow channels and water is distributed into channels with adjustable baffles. Both channels are used during high

Figure 5-10. Left: grit chamber after the bar screens (CW Slavošovice, Czech Republic, photo by Lenka Kröpfelová), right: grit chamber preceding the screens (CW Kámen, Czech Republic, photo by Jan Vymazal).

Figure 5-11. Two-chamber grit removal unit. Left: CW Ostrolovský Újezd, Czech Republic, no flow regulation; right: CW Beja, Portugal with adjustable baffles enabling the flow direction to one or both channels. Photos by Lenka Kröpfelová.

flows, one channel is used in case of low flows or in case one chamber is emptied. Disadvantage of this type is the fluctuation in flow velocity which affects the removal effect due to change in flow profile. Horizontal flow grit chambers with controlled stable flow velocity have special weirs which raise

Figure. 5-12. Horizontal flow grit removal chamber in CW Libníč, Czech Republic. *Top left*: during the construction, note the depth of a sludge compartment, *top right*: grit chamber with screen bars after completion, *bottom left*: the view from the outflow side with the triangle trough in the left, *bottom right*: detail of the weir. Photos by Lenka Kröpfelová.

water level in the channel, thus increasing surface area which in turn keeps the flow velocity constant. The shapes of the channel and the weir are connected. The most commonly used combination are triangle trough – parabolic weir, rectangular trough – hyperbolic weir or parabolic trough – rectangular weir.

For low flows, triangular troughs with parabolic weirs is the most common combination. Sand and other grit is pushed along the trough bottom and from there it falls down through the slot in the bottom to the sludge compartment (Fig. 5-12). Average retention time under maximum flow should be longer than 30 seconds and surface hydraulic loading should be < 16 m^3 m^{-2} h^{-1}.

Very small systems sometimes use horizontal flow grit chambers with a shallow sedimentation slot in the bottom (Fig. 5-10). The grit is periodically removed manually by shovel and stored temporarily in the side compartment (Fig. 5-13). However, the volume of these compartments is rather small and the storage compartments need to be cleaned after every major storm event.

Grit composition is highly variable with moisture content ranging from 13 to 65%, and volatile content from 1 to 56%. The quantity of grit varies greatly among system between 0.004 and 0.22 m^3 per 1,000 m^3 (Crites and Tchobanoglous, 1998).

Figure 5-13. Horizontal flow grit chamber with side grit compartment. CW Sklené, Czech Republic. Photo by Lenka Kröpfelová.

Pretreatment units for secondary treatment constructed wetlands are most commonly septic and Imhoff tanks. Septic tanks are used to receive the wastewater discharged from individual houses but also could be used to serve clusters of homes and small communities.

The effective compartment volume of a septic tank V (m^3) is calculated according to the formula:

$$V = a\,n\,q\,t \tag{5.1}$$

where a = coefficient for sludge volume (usually a = 1.5)
n = number of connected people
q = specific water consumption ($m^3\,d^{-1}$)
t = mean detention time (usually t = 3 d)

A total septic tank volume is calculated according to the mean detention time in the effective septic tank compartment and necessary sludge compartment volume. Detention time in the septic effective compartment is commonly 3 days. For the sludge compartment it is necessary to calculate an additional 50-60% of the effective compartment volume. Septic tanks must be regularly emptied and about 15 cm of sludge must be left as inoculum.

An Imhoff tank (Fig. 5-14) consists of a two-storey tank in which sedimentation is accomplished in the upper compartment and digestion of the settled solids is accomplished in the lower compartment (Crites and Tchobanoglous, 1998). Although sludge digestion chamber can be designed to hold sludge for only a few days, it is generally designed for much greater capacity. It is given sufficient capacity to hold the sludge until it is well

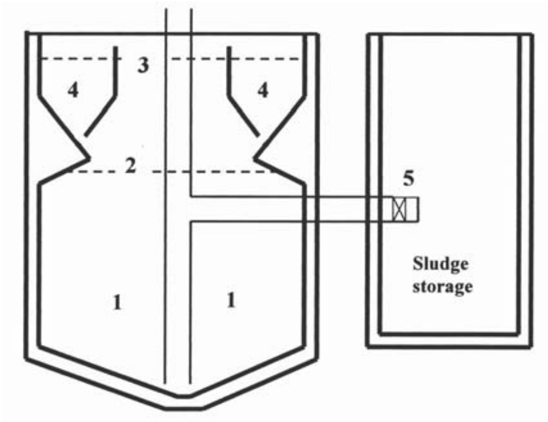

Figure 5-14. Imhoff tank. 1-digestion compartment, 2-maximum sludge level, 3-watewater level in sedimentation compartment, 4-sedimentation chambers, 5-sludge pipe (sludge could be pumped directly without storage). Drawing provided by EKOS-Hrnčíř.

digested. Digested sludge may be transferred to sludge storage tank, sludge beds or lagoons before final disposal. Imhoff tank is the most commonly-used pretreatment for small communities (Fig. 5-15).

Figure 5-15. Examples of pretreatment for secondary treatment HF constructed wetlands for 100 PE (top left, Petrovice, Czech Republic), 200 PE (top right, Skuhrov, Czech Republic) and 500 PE (bottom, Dolní Město, Czech Republic). All systems consist of screens, horizontal grit chamber and Imhoff tank. Photos by Lenka Kröpfelová.

5.2.2 Sealing the bed

HF constructed wetlands are usually sealed by an impermeable barrier in order to prevent seepage of wastewater to underground water and to ensure the controlable outflow. The most commonly used barrier are plastic liners (Fig. 5-l6) such as low or high density polyethylene or PVC. The thickness of the liner usually varies between 0.5 and 1.5 mm (Vymazal et al., 1998a).

Figure 5-16. Left: plastic liner at the experimental HF constructed wetlandat Třeboň, Czech Republic. Right: Plastic liner in between two layers of geotextile (CW Mořina, Czech Republic). Photos by Jan Vymazal.

Cooper et al. (1996) also reported the use of bentonite and the geotextile. The European guidelines (Cooper, 1990) advised that if the local soil had hydraulic conductivity of 10^{-8} m s^{-1} or less then it was likely that it contained a high clay content and could be "puddled" to provide adequate sealing for the bed.

5.2.3 Filtration materials

The media in HF constructed wetlands perform several functions; they 1) are rooting material for vegetation, 2) help to evenly distribute/collect flow at the inlet/outlet, 3) provide surface area for microbial growth, and 4) filter and trap particles (US EPA, 2000).

5.2.3.1 Inflow and outflow zones

Inflow and outflow zones (Figs. 5-17) are filled with large stones in order to provide good water distribution along the inflow zone and good even collection of water along the outflow zone (for inflow and outflow zones see also pictures in section 5.2.4.1). Evenly-graded stones in the range 50 to 200 mm can be used (Cooper, 1990). Wire mesh gabions (Fig. 5-18) are useful to retain the inlet zone stones in position whilst the media is being placed in the bed. The inlet zone stones serve as a secondary distribution allow the level of water across the bed to be equalized.

Figure 5-17. Inflow and outflow zones in CW Obecnice, Czech Republic. Photos by Lenka Kröpfelová.

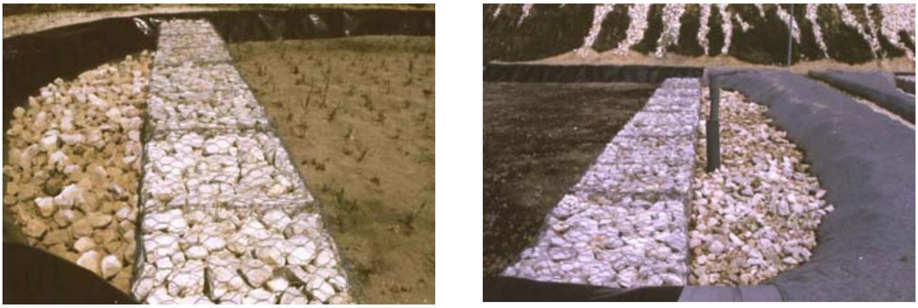

Figure 5-18. Mesh gabions in the inflow (left) and outflow (right) zones in HF constructed wetland near Leiria, Portugal. Photos by Jan Vymazal.

5.2.3.2 Filtration beds

Filtration materials may differ in their hydraulic conductivity (k_f) by several orders of magnitude. For example, coarse gravel on one end and fine-particulate clay on the other end of the spectra may have $k_f = 1$ m s^{-1} and 10^{-8} m s^{-1}, respectively (Bucksteeg, 1986; Cooper, 1990). In Seidel's hydrobotanical systems, coarse material (usually gravel or sand) were used as filtration materials. On the other hand, HF systems designed by Kickuth were filled with heavy cohesive soils with high clay content. According to Kickuth's theory the root formation would increase the hydraulic conductivity (k_f) of the soil matrix from about 10^{-5} m s^{-1} (0.86 m d^{-1}) to 10^{-3} m s^{-1} (86 m d^{-1}) within two to four years of operation (Cooper and Boon, 1987). This claim has been disputed by Bucksteeg (1985) on the basis of calculations he has made using Kickuth's assumptions and on the data available from some existing works. Further investigations in Austria, Germany, Denmark and United Kingdom in mid and late 1980s showed the claim that hydraulic conductivity would improve during the years of operation did not hold true. Instead, the hydraulic conductivity either remained stable or decreased with time, remaining on the order of 10^{-5} m s^{-1} (Brix and Schierup, 1989a; Haberl and Perfler, 1990; Netter and Bischofsberger, 1990; Findlater et al., 1990; Coombes, 1990; Bucksteeg, 1990). This resulted in in surface flow leading to channelling and scouring of the surface which resulted in areas of the bed being starved of water and this in turn led to poor reed growth. It also led to by-passing and poor treatment (Cooper and Green, 1998). Similar problems occurred with soil-based systems built in Germany and Denmark.

As a result of these problems, Water Research Centre in the UK decided in 1986/87 to recommend the use of gravel in systems at Little Stretton and Gravesend, since this would allow through-flow of water from the start. It was postulated that if the gravel beds filtered out solids and started to block the voids this might be counter-balanced by the roots and rhizomes opening up the bed. This change was very successful and more than 1,000 gravel beds have been built. The possible gravel sizes 3-6, 5-10 and 6-12 mm were recommended and the fraction 5-10 mm was recommended by European guidelines (Cooper, 1990). The gravels were washed river gravel but broken limestone has also been used successfully (Cooper and Green, 1998). Also, at the same time a flat surface was recommended because a surface slope may encourage surface-flow as happened in most systems previously built in Germany, Denmark and some early UK systems (Cooper et al., 1996).

Since then, coarse materials have successfully been used around the world but differences exist among countries and also within one country as well. The practice in Germany is to use relatively fine sand (0.2 – 1.0 mm, ATV, 1998) because it is thought to be the best compromise between available surface are for biofilm growth, suitability as a rooting medium, and hydraulic conductivity (Geller, 1996). German guidelines also recommend k_f between 10^{-3} and 10^{-4} m s^{-1} and coefficient of uniformity UC (d_{60}/d_{10})* < 5.

Austrian standards (ÖNORM, 1997, 2005) recommend the use of gravel 4-8 mm in diameter and k_f of $2x10^{-3}$ to 10^{-4} m s^{-1}. In the Czech Republic, gravel and crushed rock 4-8 mm (Fig. 5-19) is commonly used, but fraction 8-16 mm (Fig. 5-19) is often preferred because it has been shown that this fraction provides sufficient hydraulic conductivity while supporting a healthy macrophyte growth and good treatment efficincy (Vymazal, 2006a). In Denmark, the substrate must be uniform sand with a d_{10} between 0.3 and 2 mm, d_{60} between 0.5 and 8 mm, and the uniformity coefficient should be < 4

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*d_{10} and d_{60} are grain diameters in mm at which 10% and 60% of the particles are finer by weight

(Ministry of Environment and Energy, 1999). In Italy, gravel with grain size between 4 and 16 mm are commonly used (Pucci et al., 2004). Velayos et al. (2006) reported that

Figure 5-19. Washed gravel 4-8 mm in CW Sedlce, Czech Republic (top) and crushed rock 16-32 mm in CW Mořina, Czech Republic (bottom). As a scale, coin with a 20 mm diameter is used. Photos by Lenka Kröpfelová and Jan Vymazal.

most HF constructed wetland in Catalonia, Spain are designed with filtration medium with a diameter between 2 and 20 mm.

Very variable size of gravel media has been reported by various authors in the North America. In 1988, the U.S. EPA proposed media sizes ranged from d_{10} of 1 to 8 mm, with associated hydraulic conductivities of 4.9 x 10^{-3} to 5.8x10^{-3} m s^{-1}. The new version of U.S. EPA manual (U.S. EPA, 2000) recommended gravel size between 20 and 30 mm. For the first 30% of the wetland cell no more than 1% of the clean bed hydraulic conductivity is used for design while for final 70% of the wetland cell 10% of the clean bed hydraulic conductivity is recommended (Fig. 5-20). The Tennessee Valley Authority guidelines (Steiner and Watson, 1993) recommended a gravel medium of 3 to 6 mm and a hydraulic conductivity of 3x10^{-3} m s^{-1} to be used to account for clogging in the gravel bed. In 1993, U.S. EPA Technology Assessment (Reed, 1993) media sizes between 12 and 25 mm with hydraulic conductivity of 1.2x10^{-3} m s^{-1} were recommended. Reed et al. (1995) recommended a wide range of media sizes between 2 and 128 mm, with hydraulic conductivities between 1.2x10^{-3} m s^{-1} and 2.8 m s^{-1}.

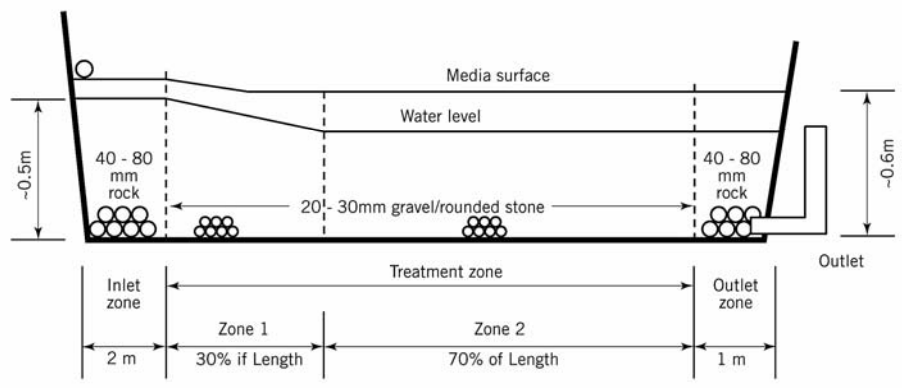


Figure 5-20. Hydraulic conductivity zones within a HF constructed wetland proposed by U.S. EPA (U.S. EPA, 2000, with permission).

The bed will not maintain the clean media conductivity because of the deposition of solids and the blockage of pore space by plant roots. If one third of the pore space is blocked, the hydraulic conductivity will decrease by one order of magnitude. This possibility must be acknowledged in design if the potential for flooding (surfacing) is to be minimized (Kadlec and Knight, 1996). In Table 5-2 "clean" and "dirty" values of hydraulic conductivity reported in the literature. For other k_f values see, for example Hu et al. (1994) or Wood (1995).

Table 5-2. Hydraulic conductivity (k_f) values reported in the literature. NR = not recorded. (U.S. EPA, 2000, with permission).

Size and type of media	"Clean"/"Dirty" k_f (m d^{-1})	Type of wastewater	Length of operation	Notes
5-10 mm gravel [1]	34,000/12,000	Secondary	2 years	Downstream portion
5-10 mm gravel [1]	34,000/9,000	Secondary	2 years	Inlet zone
5 mm gravel[2]	6,200/600	Landfill leachate	26 months	
30-40 mm gravel [3]	NR/1,000	Secondary	2 years	First 6 m of the bed
5-14 mm gravel [3]	NR/12,000	Secondary	2 years	Last 9 m of the bed
19 mm rock[4]	120,000/3,000	Septic tank	7 months	

[1]Bavor et al. (1989), Fisher (1990), Bavor and Schulz (1993); [2]Sanford et al., (1995a,b), Sanford (1999), Surface et al. (1993); [3] Sapkota and Bavor (1994); [4]George et al. (2000)

5.2.3.3 Clogging

Some HF constructed wetlands have experienced conditions called "surfacing" where a portion of the wastewater flows on top of the media. Surfacing 1) creates conditions favorable for odors and mosquito breeding,

2) creates a potential health hazard for persons and animals that may come into contact with the wastewater, and 3) reduces the hydraulic retention time in the system (US EPA, 2000). However, it seems that surfacing does not hamper substantially the treatment performance of the system – it fact, it creates a combination of FWS and HF systems. The early HF systems built in Germany and Denmark suffered from surfacing as a consequence of soil filtration material clogging but the treatment performance of the systems was still very good (e.g., Brix, 1998).

Surfacing (or ponding) occurs whenever the hydraulic conductivity of the media is not sufficient to transport the desired flow within the usable headloss of the media. The usable headloss is defined by the difference in the elevations of the outlet piping and the top of the media. Surfacing can result from a number of factors including 1) poor design of the system inlet and outlet piping, 2) inaccurate estimate of the clean hydraulic conductivity of the media, 3) improper construction, 4) inaccurate estimate of the reduction in hydraulic conductivity, or 5) clogging, that will occur due to growth of plant roots, solids accumulation and biomat formation (biomat is a term used mostly in the United States for slimes and sludges containing microbial biomass). Several researchers have found that clogging was the most severe within the first 1/4 to 1/3 of the beds (Bavor et al., 1989; Fisher, 1990; Tanner and Sukias, 1995; Tanner et al., 1998b; Young et al., 2000; George et al., 2000, see also Fig. 5-21). The hydraulic conductivity was found to be less restricted and fairly uniform over the remaining length of the system. However, Cooper et al. (2005) reported that during the survey between November 2002 and June 2004 in the United Kingdom, 21 out of 126 surveyed tertiary HF systems suffered from water on most of bed surface. Cooper et al. (2007) updated this information and they reported surfacing on most of 76 tertiary beds out of 255 surveyed systems. The surfacing, however, did not deteriorate the treatment effect of the systems.

Figure 5-21. Surfacing (ponding) of wastewater in the inlet zone of HF constructed wetlands Břehov (left) and Mořina (right), Czech Republic. Photo by Lenka Kröpfelová.

The early statement by Kickuth (1981) that plant roots significantly increase the hydraulic conductivity of the filtration media by opening up preferential pathways for the wastewater flow proved to be wrong. Based on recent studies, the presence of plant roots and rhizomes in the HF beds will have a negative effect on hydraulic conductivity (Sanford et al., 1995a,b; Breen and Chick, 1995; George et al., 2000; Young et al., 2000). However, most of the hydraulic conductivity decline is apparently associated with sediments and biomat formation since unplanted gravel beds show declines similar to those found in planted systems (e.g., Fisher, 1990).

Wastewater entering the filtration beds of constructed wetlands always contains suspended solids which are trapped within the filtration material. This entrappment is the highest within the area where distribution zone filled with large stones meets with the filtration bed material which has much smaller grain size. Within the filtration bed, also precipitation, such as sulfide formation, may contribute to bed clogging (Liebowitz et al., 2000).

Microbial biofilms, which are active in removal of dissolved pollutants, may have a negative impact on hydraulic conductivity and can contribute to bed clogging. The microbial biofilms form in response to both particulate and soluble organic loadings (Wallace and Knight, 2006). These biofilms entrap both organic and inorganic solids (Winter and Goetz, 2003), forming a biomat. Biomat formation is greatest at the inlet zone, usually several meters, where the organic loading is highest (Ragusa et al., 2004). The loss of pore volume due to biomat formation reduces the hydraulic conductivity in this inlet zone (Zhao et al., 2004a). Organic matter is removed as wastewater flows through the wetland, resulting in declining biomat formation. At the outlet, where only small quantities of organic matter are available to the microbial biofilm, biomat formation is minimal (Figs. 5-22, 5-23).

Figure 5-22. HF system at Břehov, Czech Republic. Gravel substrate taken 2 meters (left) and 25 meters (right) from the inflow. Note substantial biomat formation in the inlet sample. Photos by Lenka Kröpfelová.

In fine-grained materials, there is greater surface area available per unit length of flow path. As a result, more microbial biofilm can form in response to the organic loading. Because the pore size is smaller, these biofilm are more effective in entrapping organic and inorganic solids. If the resulting

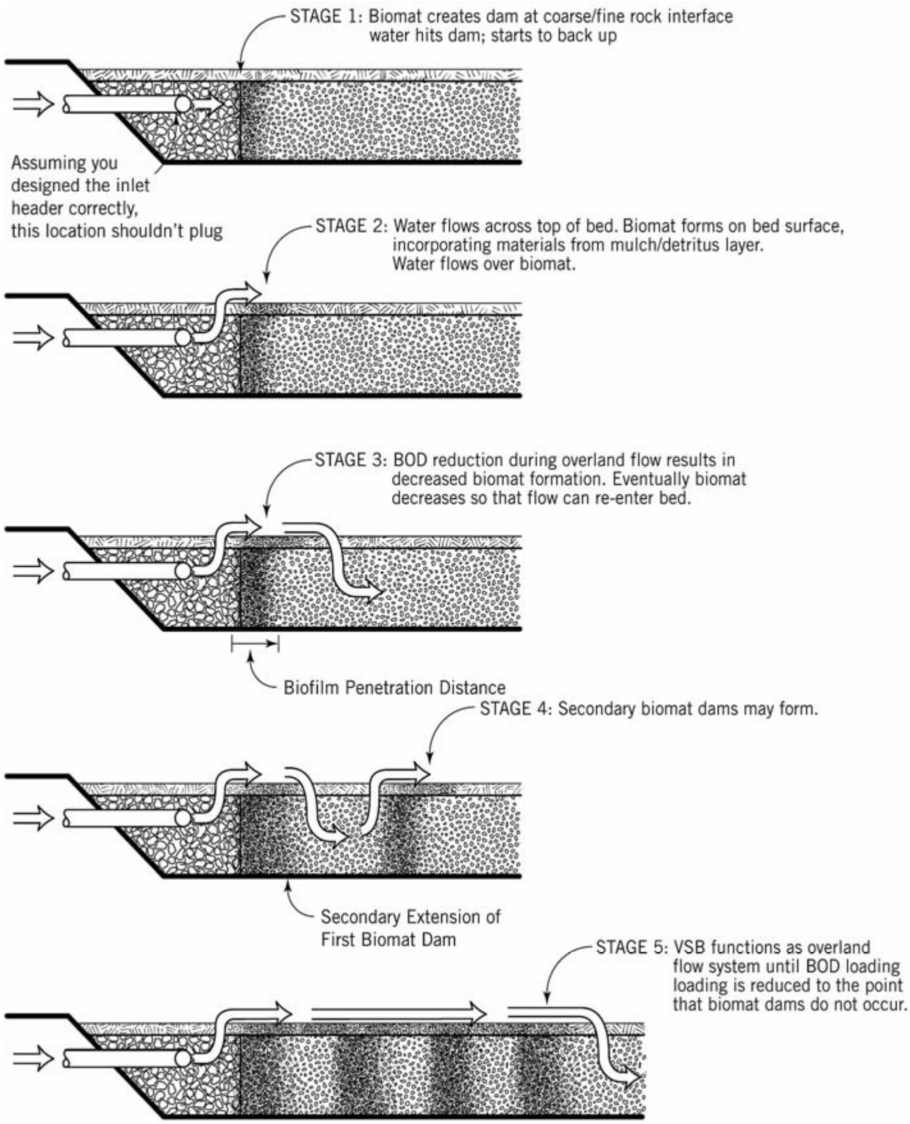


Figure 5-23. Stages of clogging in HF (VSB) constructed wetlands. From Wallace and Knight (2006). Reprinted with permission from the Water Environment Research Foundation: Alexandria, Virginia.

biomat completely fills the pore spaces, the hydraulic conductivity is controlled by the characteristics of the biomat and not by the characteristics of the media, and the ponding will likely occur. In coarse materials, there is less surface area available for biofilm formation per unit length of flow path. As a result, less microbial biofilm can form in response to the organic loading. Due to the larger pore spaces, the biomat cannot completely fill the pore volume, and effective flow paths through the media still exist. The net effect lengthens the biomat penetration distance but decreases the potential for overland flow (Wallace and Knight, 2006).

The current understanding of HF CWs design does not allow a quantitative determination of the biomat penetration distance (Wallace and Knight, 2006). Bavor et al. (1989) noted that clogging within a series of very long, narrow HF cells was remedied by excavating the first 5 m of the bed and replacing with coarse rock. Watson et al. (1989) provided evidence from the Benton, Kentucky system that the farthest clogged reach penetrated 100 m into a 300 m bed.

The clogging mat within the pores consists of anaerobic decomposition products such as polysaccharides and polyuronides (Thomas et al., 1966) and/or unaltered organics when low temperatures retard decomposition (De Vries, 1972). Siegrist (1987) developed an empirical relationship between infiltration capacity as a function of time and the cumulative density loadings of total BOD_5 and TSS. The model of sand clogging by suspended solids was described by Blažejewski and Murat-Blažejewska (1997).

5.2.4 Water distribution and collection

5.2.4.1 Water distribution

In HF constructed wetlands the aim is to get even distribution across the full-sectional area of the inlet end of the bed. The early systems used long channels with castellated or vee-notch weirs (Fig. 5-24). However, they had problems with screenable material collected at edges and causing maldistribution. Also, these were expensive to construct and so there was a tendency to move to single pipes with adjustable tees (Fig. 5-24) or orifices spread along their length (Cooper et al., 1996). The orifices should be evenly spaced at a distance approximately equal to 10% of the cell width. For example, a system 20 m wide should have orifices placed every 2 m. If poor design causes wastewater to always discharge through only some orifices, clogging of the media or accumulation of a surface layer of solids near those orifices can become a problem, especially for an influent with relatively high suspended solids (US EPA, 2000).

During the mid-1990s, a buried pipe with upward pointing riser pipes (each fitted with a vee-notch) has been used widely in Severn Trent Water, United Kingdom. These riser pipes were fitted with sliding collars which allow the flow to be adjusted (Fig. 5-25).

At present, the most common way of water distribution is the use of perforated plastic pipes, usually made of reinforced polyethylene. The holes in the pipes must be large enough, at least 3 cm in diameter to prevent uneven water distribution due to hole clogging. There are several options how to place the distribution pipe within the inflow zone. The most common placements are: 1) pipe is buried in the inflow zone and the surface of the inflow zone is leveled with filtration bed (Fig. 5-26); 2) pipe is laid down on the surface of the inflow zone and it is not covered (Fig. 5-27); 3) pipe is laid on the inflow zone surface and it is overlaid with large stones (Fig. 5-28).

Figure 5-24. Open inflow trench in HF CW Acle, United Kingdom (left) and adjustable tees in HF CW Otorohanga, New Zealand (right). Photos by Jan Vymazal.

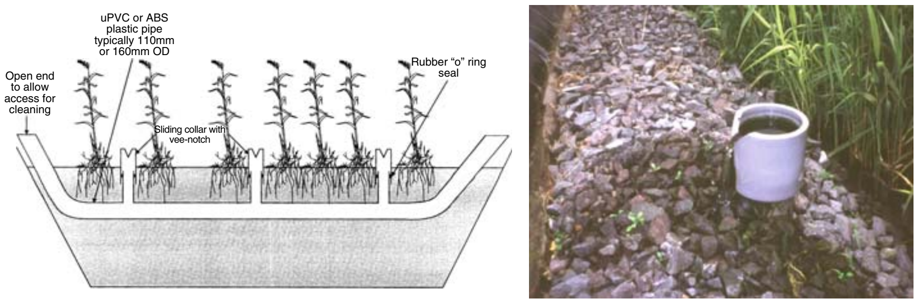


Figure 5-25. Riser pipe distribution. Left: general layout, with permission from Backhuys Publishers, right: detail of the slide collar with a vee-notch, HF Moreton Morrell. Photo by Jan Vymazal.

Figure 5-26. Distribution pipe buried on the bottom of the distribution zone. HF CW Čejkovice, Czech Republic. Left: construction of the inflow zone, right: Inflow zone after completion and in operation. Photos by Jan Vymazal.

Figure 5-27. Distribution pipes on the surface of distribution zone. HF CW Čehovice, Czech Republic. Left: general view, right: detail of the wastewater inflow. Photos by Lenka Kröpfelová.

Figure 5-28. Distribution pipe laid down on the surface and overlaid by large stones. HF CW Mořina, Czech Republic. Left: March 2006 before plants start to sprout, right: May 2006. Photos by Jan Vymazal.

The distribution pipes are commonly furnished with upright pipes at both ends of the inflow zone which enable pipe cleaning (Fig. 5-29). However, if pipes are too long it is not possible to suck out the sediment and the accumulated solids need to be flushed out. This is not an ideal situation as these particles will contribute to bed clogging. Also, the upright pipes enables the inspection of water level in the inflow zone.

Figure 5-29. Inspection pipes at the end of inflow zone. Left: CW Čejkovice, right: Mořina, Czech Republic. Photos by Jan Vymazal.

Figure 5-30. Deep zone distribution of wastewater in HF CW Ondřejov, Czech Republic. Photo by Jan Vymazal.

Distribution may also be realized by simple deeper zone within the inflow zone (Fig. 5-30). However, it is necessary to make sure that wastewater enters the filtration bed along the entire length of the inlet zone and there is no just one outlet to the bed (Fig. 5-31). Of course, there are other options for wastewater distribution due to creativity of designers (see e.g., Fig. 7-36).


Figure 5-31. Distribution of wastewater in HF CW Břehov, Czech Republic. Left: Wastewater flows through the middle of the bed causing shortcircuiting, right: raised wall which spreads the wastewater along the whole inlet zone. Photos by Lenka Kröpfelová.

5.2.4.2 Water collection and outflow structures

At the outlet end most systems usually have a perforated agricultural drain pipe buried on the bottom of the outflow zone filled with stones (see Figs. 4-40 and 5-32). This leads to a sump where the water level is controlled by either a socketed pipe (Fig. 5-33), swivelling elbow (Fig. 5-34) or flexible hose(s) (Fig. 5-35) which can be held in position by a chain or rope. At present, swivelling or rotating elbows are usually made of plastic. However, the early outflow structures were quite often equipped with steel


Figure 5-32. Outflow zones are leveled with the filtration bed surface. The zones are frequently equipped with inspection pipes. Left: CW Sklené, Czech Republic, right: CW Mořina, Czech Republic. Photos by Jan Vymazal.

elbows which suffer from rusting and it was necessary to change them frequently. Long outflow zones may have more than one outflow structure (Fig. 5-36).

Figure 5-33. Socketted pipe at the outflow from a HF CW in Krásna Lúka, Slovakia, Photo by Jan Vymazal.

Figure 5-34. Swivelling elbows hanging on a chain in CW Sklené, Czech Republic(left) and Waikeria, New Zealand (right). Photos by Jan Vymazal.

Figure 5-35. Flexible hoses at the outflow from a HF CW Mořina, Czech Republic. Photo by Jan Vymazal.

Figure 5-36. Wide outflow zones may be equipped with a series of outflow structures in order to provide an even discharge along the outflow zone. Left: Yaintian, China, right: Piracicaba, Brazil. Photos by Jan Vymazal.

5.2.5 Vegetation

The emergent macrophytes growing in horizontal flow constructed wetlands designed for wastewater treatment have several properties in relation to the treatment processes that make them an essential component of the design (see section 3.5). The most important effects of the emergent macrophytes in relation to the wastewater treatment processes in constructed wetlands with horizontal sub-surface flow (HF CWs) are the physical effects such as erosion control, provision of surface area for attached microorganisms and insulation of the bed surface during winter (Brix, 1997, 2003; Vymazal et al. 1998a).

The metabolism of the macrophytes (e.g., plant nutrient uptake, oxygen release, release or organics) affects the treatment processes to different extents depending on design but in HF CWs those processes play less important role than the physical processes (Brix, 1993b, 1997, 1998, 2003a; Brix and Schierup, 1990; Vymazal 1999a, 2001b, 2004 a,b). However, in the tropics and subtropics with no dormancy season removal of nutrients via continuous harvesting can be significant removal mechanism (e.g., Okurut, 2001). Decomposing wetland plants and plant root exudates are potential sources of biodegradable organic carbon for denitrification but are also sources of organic nitrogen, which is easily converted to ammonia (US EPA, 2000).

It has been suggested that in temperate and cold climates insulation of the bed during cold periods is probably the most important role of vegetation in HF CWs (Mander and Jenssen, 2003; Vymazal and Kröpfelová, 2005) and therefore, it is desirable to use plants which have high aboveground biomass and grow fast and create a dense cover soon after planting. The plants used in constructed wetlands designed for wastewater treatment should also 1) be tolerant to high organic and nutrient loadings, 2) have rich belowground organs (i.e., roots and rhizomes) even under certain level of anoxia and/or anaerobiosis in the rhizosphere in order to provide substrate for attached bacteria and oxygenation (even very limited) of areas adjacent to roots and rhizomes (Čížková-Končalová et al., 1996b; Květ et al., 1999).

5.2.5.1 *Phragmites* spp.

By far, the most commonly used plant for HF constructed wetlands is *Phragmites australis* (Cav.) Trin. ex Steudel (Poaceae) (Common reed) (Fig. 5-37). *P. australis* (= *Phragmites communis* Trin.) is a perennial and flood-tolerant grass with extensive rhizome system (Fig. 5-38) which usually penetrates to depths of about 0.6 to 1.0 m. Stems are rigid with hollow internodes (Fig. 5-38). Inflorescence is a drooping panicle up to 50 cm long (Fig. 5-39). A number of morphological variants of the species have been recorded which range in shoot height from 0.5 m to giant forms about 8 m tall from the marshes of the Danube River (Haslam 1971a,b) and Tigris and

Euphrates Rivers (Maxwell, 1957; Thesinger, 1964). Reproduction is from seeds and from the rhizome or creeping stolons. However, despite producing large quantities of seeds, germination rates are generally very low (Harris and Marshall, 1960; Galinato and van der Valk, 1986; Tucker, 1990; Marks et al., 1994).

Common reed is a cosmopolitan grass occurring as a dominant component in the freshwater, brackish and in some cases also marine littoral communities almost all over the world (Dykyjová and Hradecká, 1976; Hocking et al., 1983; Soetaert et al., 2004). It often occurs in large monospecific stands (Pénzes, 1960; Hocking et al., 1983). Its distribution is widespread throughout Europe, Africa, Asia, Australia and North America between 10 and 70^{o} latitude (Hawke and José, 1996). Although normally associated with lowlands, it has been recorded at 3,000 m in Tibet (Crook et al., 1983).

P. australis has wide ecological amplitude (Haslam, 1973a; Rodewald-Rudescu, 1974). The habitat description range from oligotrophic lakes in Sweden (Björk, 1967) to heavily polluted lakes (e.g., Dykyjová and Hradecká, 1973). In Europe, *P. australis* is generally regarded as an ecologically beneficial plant providing habitat for endangered wildlife (Nevel et al., 1997), buffer zones for nutrient retention (Toth, 1972) and stabilizing shore banks (Rolletschek, 1997). Additionally, *P. australis* is of economical importance because reeds may be harvested for roof thatch, fence material, fuel etc. (Haslam, 1973b; Granéli, 1984). However, in North America, New Zealand and and some parts of Australia *Phragmites* is considered as invasive introduced pest species (Roman et al., 1984; Hocking, 1989; Marks et al., 1994; Meyerson et al., 2000).

Zemlin et al. (2000) pointed out that the common reed shows marked differences in stand structure and morphology both among years and among sites (Björk, 1967; Haslam, 1971c, Dykyjová and Hradecká, 1973; Rodewald-Rudescu, 1974). Differences between stands have often been related to the trophic state of their sites (Rodewald-Rudescu, 1974; Bornkamm and Raghi-Atri, 1986; Kühl and Kohl, 1993; Kohl et al., 1998). Other investigations indicated that differences in reed morphology and performance have a genetic component (Daniels, 1991; Kühl et al., 1997; Lippert et al., 1999).

Maximum aboveground biomass of *P. australis* is highly variable depending on latitude, climate, salinity, water depth, eutrophication and interactions between these factors (Soetaert et al., 2004). In the literature, there are numerous and highly variable data on *Phragmites* aboveground maximum biomass. For example, Čížková (1999) reported the maximum seasonal biomass from 12 natural stands in 9 European countries between 308 and 4,165 g DM m^{-2}, Vymazal and Kröpfelová (2005) in their review reported the range between 413 and 9,890 g DM m^{-2} for 12 natural stands from Europe, Asia and Australia. However, the most common values for maximum aboveground biomass found in natural stands are between 1,000 and 2,000 g DM m^{-2}.

Figure 5-37. *Phragmites australis* growing in HF constructed wetland Studénka in the Czech Republic with the maximum aboveground biomass of 11,280 g DM m^{-2} and average stem length 337 cm. Photo by Lenka Kröpfelová.

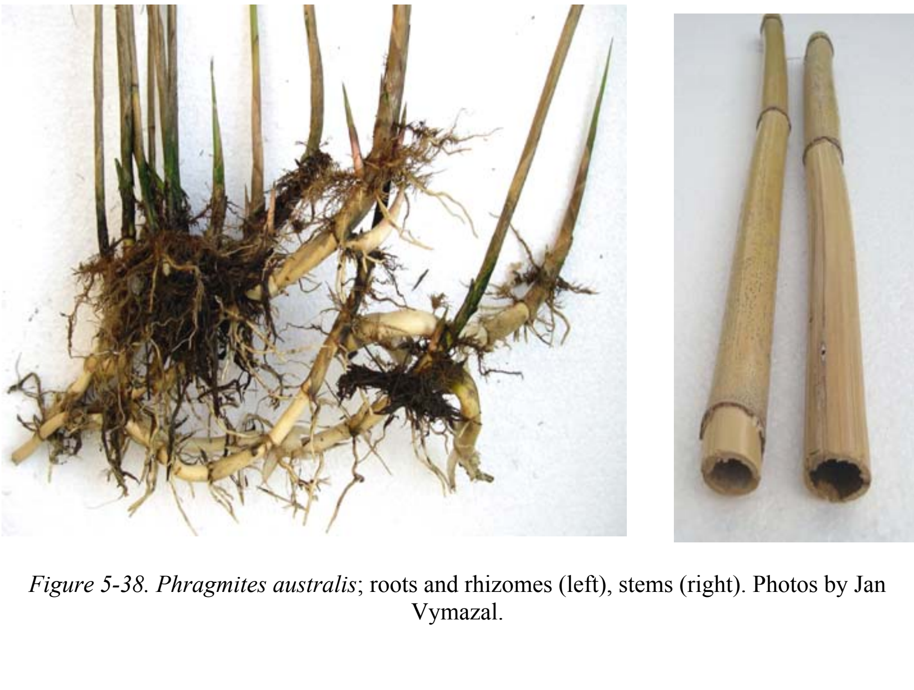


Figure 5-38. Phragmites australis; roots and rhizomes (left), stems (right). Photos by Jan Vymazal.

Figure 5-39. Phragmites australis inflorescence. HF constructed wetland Alcochete, Portugal. Photo by Lenka Kröpfelová.

The R/S (root/shoot) ratio in natural stands is usually high, indicating high underground biomass. Čížková (1999) reported an average R/S ratio of 7.5 with values ranging between 1.5 and 21.9 (see also Table 3-3). With little external loss, the maximum aboveground mass is generally assumed to be within 85-100% of net annual aboveground production (Mason and Bryant, 1975; Ondok and Květ, 1978; Gessner et al., 1996, Brix et al., 2001a).

The values of maximum aboveground biomass recorded for *Phragmites* growing in HF constructed wetlands (Table 5-3) are within the range found in natural stands but it seems that the most common values could be a bit higher in constructed wetlands. However, there is much less data from constructed wetlands to confirm this suggestion. It seems apparent that there is a difference in R/S ratios. While in natural stands belowground biomass forms much higher part of the total biomass, in constructed wetlands the belowground biomass is usually lower than aboveground (Table 5-3). This may be due to several reasons such as continuous easy availability of nutrients in the root zone or stress caused by high level of pollutants.

While *Phragmites australis* is used throughout Europe (Figs. 5-40 and 5-41, see also Figs. 4-55, 5-2, 5-3) Canada (Fig. 5-42, see also Fig. 4-56), Australia and most parts of Asia. In the United States and New Zealand, Common reed is considered as introduced and invasive species and its use is restricted or prohibited. *Phragmites mauritianus* Kunth is used in Africa (e.g., Sekiranda and Kiwanuka, 1998; Okurut et al., 1999; Okurut 2001; Byekwaso et al., 2002) (see Fig. 5-5) and *Phragmites karka* is common in India and Nepal (e.g., Billore et al., 1999; Bista et al., 2004) (see Fig. 7-52).

Table 5-3. Examples of maximum annual aboveground biomass (g DM m^{-2}) and belowground/aboveground ratio (R/S ratio) for *Phragmites australis* growing in HF constructed wetlands.

Country	**Biomass**	**R/S ratio**	**Reference**
Australia	788	2.43	Adcock and Ganf (1994)
USA, New York average from 3 beds with different substrate in HF treating landfill leachate	1,113	0.79	Peverly et al. (1995)
Germany	1,360	0.42	Gries and Garbe (1989)
Czech Republic average from 5 HF systems	2.088		Vymazal et al. (1999)
Czech Republic average from 2 HF systems	2,172		Dušek and Květ (1996)
Austria average from 3 beds with different loading	2.733	0.50	Haberl and Perfler (1990)
Czech Republic average from 13 HF systems with the biomass range of 1,652 - 5,070 g m^{-2}	3,266		Vymazal and Kröpfelová (2005)
USA, Alabama	6,065	0.78	Behrends et al. (1994)
Sicily, Italy	7,929		Barbera et al. (2007)
Czech Republic	11,280		Vymazal and Kröpfelová (2005)

Figure 5-40. HF constructed wetland Machová in the Czech Republic planted with *Phragmites australis*. Photo by Lenka Kröpfelová.


Figure 5-41. HF constructed wetland Beja, Portugal planted with *Phragmites australis*. Top: general view, bottom: detail on *Phragmites* stand. Photos by Jan Vymazal and Lenka Kröpfelová.

Figure 5-42. HF constructed wetland at Cap St. Jaques, Québec, Canada planted with *Phragmites australis*. Photo by Jacques Brisson, with permission.

5.2.5.2 *Phalaris arundinacea*

Phalaris arundinacea L. (Poaceae) (Reed canarygrass) is a 1 to 3 m tall (Fig. 5-43), long-lived perennial grass with a C3 photosynthetic pathway (Kephart and Buxton, 1993; Lewandowski et al., 2003). It produces a dense crowns and prominent networks of vigorous roots and rhizomes (Fig. 5-44), penetrating to a depth of about 30-40 cm, allowing for aggressive vegetative spread (Coops et al., 1996; Kätterer and Andrén, 1999). Panicles (Fig. 5-45) are 7 to 40 cm long, bearing wingless glumes that contain both fertile and sterile florets (Carlson et al., 1996; Lewandowski et al., 2003). Seed germination requires light and is best in moist soils (Vosse, 1962; Landgraff and Juttila, 1979, Linding-Cisneros and Zedler, 2002), with highest germination rates in water-saturated soils (Coops and van der Velde, 1995; Kellogg et al., 2003).

Lavergne and Molofsky (2004) in their excellent review on *P. arundinacea* summarized that it typically grows best under cool and moist conditions (Coops et al., 1996; Sahramaa and Jauhiainen, 2003) and it is found in a large array of wetland habitats, such as wet meadows and lake shores (e.g. Odland, 1997; Galatowitsch et al., 2000; Odland and del Moral, 2002), dynamic river banks (Coops and van der Velde, 1995; Henry and Amoros, 1996), and floodplains (Šrůtek, 1993; Klimešová, 1994, 1995). Although *P. arundinacea* is most prevalent in wet areas, it is also found on upland sites, where it can survive temporary droughts (Vose, 1959; Shaeffer and Marten, 1992; Troccoli et al., 1997). *Phalaris* grows rapidly and tends to form monocultures (Marten, 1985; Apfelbaum and Sams, 1987; Miller and Zedler, 2003). *Phalaris* is a species typical of lowland river floodplains (Kopecký, 1969), however it is found at altitudes > 1300 m (Klimešová, 1996).

Figure 5-43. Phalaris arundinacea growing in the HF constructed wetland Břehov, Czech Republic. Photos by Jan Vymazal.

Studies have shown that *Phalaris* responds positively to nutrient enrichment. The controlled experiments of Green and Galatowitsch (2001) and Maurer and Zedler (2002) showed that high nutrient treatments increased biomass of *Phalaris* and increased allocation to aboveground growth. Also Kätterer and Andrén (1999) reported that fertilization decreased the amount of biomass allocated belowground as compared to aboveground biomass. Wetzel and van der Valk (1998) found that although soil moisture did not have a significant effect on *Phalaris* productivity, their high nutrient treatment increased biomass 73% over the low nutrient treatment. *Phalaris* appears to grow best where water levels are highly variable (Cooke and Azous, 1997; Miller and Zedler, 2003).

Figure 5-44. Phalaris arundinacea – belowground biomass. Left: one month old plants, right: third growing season. Photos by Jan Vymazal.

Figure 5-45. Inflorescence of *Phalaris arundinacea*. Left: early stages of flowering, right: shortly after heading. Photos by Lenka Kröpfelová.

Reed canarygrass is native to the temperate zones of the Northern Hemisphere and is widely distributed throughout Eurasia (Lavergne and Molofsky, 2004). It was originally introduced from Europe to the United States shortly after 1850 and has since spread throughout North America (Merigliano and Lesica, 1998; Galatowitsch et al., 1999) and it is considered as invasive species especially in anthropogenically disturbed areas (Lavergne and Molofsky, 2004; Kerchler and Zedler, 2004; Perkins and Wilson, 2005).

P. arundinacea is used for variety of purposes including forage crop (e.g., Lawrence and Ashford, 1969; Marten and Hovin, 1980; Foss, 1982; Shaeffer and Marten, 1992; Buxton et al., 1998; Min et al., 2002), persistent cover for permanent pastures (e.g., Hoveland, 1992; Casler et al., 1998), phyto-extraction of soil contaminants (e.g., Lasat et al., 1997; Samecka-Cymerman and Kempers, 2001), re-vegetation and stabilization of shorelines and river banks (e.g., Holmberg, 1959; Casler and Hovin, 1980; Conchou and Fustec, 1988; Figiel et al., 1995), bioenergy crop (e.g., Burvall, 1997; Mattsson, 1997; Hadders and Olsson, 1997; Nilsson and Hansson, 2001; Lewandowski et al., 2003), paper, pulp and fibre production (e.g., Wisur et al., 1993; Kätterer et al., 1998; Pahkala and Pihala, 2000; Saijonkari-Pahkala, 2001; Finell et al., 2002; Hellquist et al., 2003).

Maximum aboveground biomass in natural stands is variable depending mostly on the trophic status of the stand. Vymazal (2005a) in his review reported values between 440 and 2304 g DM m^{-2} with the lowest value recorded in a meso-eutrophic lake in Scotland (Ho, 1979a) and the highest value recorded for wet meadows in the Czech Republic (Hlávková-Kumnacká, 1980). Sparse results from HF constructed wetlands indicate that *P. arundinacea* maximum annual standing crop is within the range found in natural stands (Table 5-4). The values in Table 3-2 indicate that maximum standing crop in fact nearly equals the net annual productivity. In natural stands, the R/S ratio for *Phalaris arundinacea* is usually between 1 and 2. In constructed wetlands, however, the R/S ratio is usually < 1.0 (Table 5-4).

Table 5-4. Examples of maximum annual aboveground biomass (g DM m^{-2}) and belowground/aboveground ratio (R/S ratio) for *Phalaris arundinacea* growing in HF constructed wetlands.

Locality	Biomass	R/S ratio	Reference
Czech Republic[1]	731		Vymazal et al. (1999)
USA, Alabama	831	0.75	Behrends et al. (1994)
Czech Republic[2]	1,286		Vymazal and Kröpfelová (2005)
USA, New York[3]	1,713	0.23	Bernard and Lauve (1995)
Czech Republic	1,780	0.36	Unpublished results
Czech Republic	2,265	0.54	Unpublished results

[1]Average value for two systems, [2]average value for 7 systems sampled in 2003 ranging from 305 to 1,902 g DM m^{-2}, [3]treatment of landfill leachate, [4]CW Břehov, [5]CW Mořina.

P. arundinacea has commonly been used for HF constructed wetlands in the Czech Republic either as single species (Fig. 5-46) or in combination with *Phragmites australis* (Fig. 5-46). The use of reed canarygrass was also reported from the United States (e.g., Behrends et al., 1994; Bernard and Lauve, 1995).

Figure 5-46. Top: HF constructed wetland Čejkovice, Czech Republic planted with *Phalaris arundinacea* as single species. Bottom: HF constructed wetland Příbraz, Czech Republic planted with *Phalaris arundinacea* (at left) and *Phragmites australis* in bands perpendicular to flow path. Photos by Lenka Kröpfelová.

5.2.5.3 *Typha* spp.

Typha spp. (Cattails) (Typhaceae) (Figs. 5-6, 5-47) are erect rhizomatous perennial plants with jointless stems. Rhizomes are extensive branched, produce aerial shoots at intervals (Fig. 5-48) and grow in shallow depth in horizontal direction. Leaves are flat to slightly rounded and obtain height up to 3 m. Inflorescence (Fig. 5-49) is a densely compact cylindrical, 15-50 cm long spike, which can produce up to 200,000 seeds with a high percentage of viability (Prunster, 1940; Yeo, 1964). The female spikes are below the males and the distance between them serves as the basis for determination.

Figure 5-47. Typha angustifolia growing in HF constructed wetland Veselý Žďár, Czech Republic. Photo by Jan Vymazal.

Figure 5-48. Roots and rhizomes of *Typha latifolia*. Photo by Lenka Kröpfelová.

Figure 5-49. Inflorescences of *Typha latifolia* (left) and *Typha angustifolia* (right). Photos by Jan Vymazal.

Cattail species are commonly found inhibiting shallow bays, irrigation ditches, lakes, ponds, rivers and both brackish and fresh water marshes.There are four major *Typha* species, among others, found in wetlands: *Typha latifolia* L. (Common cattail, Broad-leaved cattail), *Typha angustifolia* L. (Narrow-leaved cattail), *Typha domingensis* Pers. (Southern cattail, Santo Domingo cattail), *Typha glauca* Godr. (Blue cattail). *T. latifolia* and *T. angustifolia* are cosmopolitan species, *T. latifolia* is not found in central and south Africa. *T. glauca*, a hybrid of *T. latifolia* and *T. angustifolia*, is most common in North America and *T. domingesis* is found in subtropical and tropical parts of Americas, Australia and Africa.

The most common *Typha* species – *T. latifolia* nd *T. angustifolia* could be distinguished according to the spike position. While in *T. latifolia*, there is no space between male and female spikes, in *T. angustifolia* both spikes are separated (Fig. 5-49). These two species also differ in environmental requirements. *T. angustifolia* prefers water depth between 30 and 110 cm and it copes well with short-term decrease of water level and even drought. On the other hand, *T. latifolia* prefers water depth between 20 and 50 cm and does not like water level fluctuations.

Typha plants are one of the most famous edible and useful plants in the world, often referred to as “supermarket of the marsh”. The young leaves are excellent salad greens and can be cooked as potherbs before the pollen spike is ripe, it can be boiled and eaten much like corn on the cob (Reddington, 1994). However, at present the value as food source is generally negligible. Cattail leaves can be used for production of mats, chairs or baskets. *Typha* seeds could also mixed with clay to produce bricks with good insulation properties (Mauring et al., 2003).

Typha is a very productive species, (see Table 3-2) with maximum aboveground biomass values found in natural stands exceeding 5,000 g DM m^{-2}. In constructed wetlands, *Typha* plants are usually used in free water surface systems (e.g., Herskowitz, 1986; Maddison et al., 2003; Obarska-Pempkowiak and Ozimek, 2003; Maine et al., 2006) where maximum aboveground biomass is comparable with natural stands. Vymazal (2005a) in his review reported the maximum aboveground biomass values from 7 FWS systems between 592 and 5,602 g DM m^{-2}. The use of *Typha* in HF constructed wetlands is limited mainly because the underground structures (roots and rhizomes) are very shallow, sediments occupied by *Typha* are usually more anaerobic then in the presence of other plants and also the rate of humification, i.e. creation of soil layer within the root zone, is much faster as compared to other species.

Examples of HF constructed wetlands planted with *Typha* sp. are shown in Figures 5-47 and 5-50.

Figure 5-50. HF constructed wetland NATIVA I in Uruguay planted with *Typha domingensis*. Photo by Silvana Perdomo, with permission.

5.2.5.4 *Glyceria maxima*

Glyceria maxima (Hartm.) Holmb. (Poaceae) (Sweet mannagrass, Reed sweetgrass, Giant sweetgrass) (= *Glyceria aquatica* (L.) Wahlenb.) is a robust perennial grass up to 2.5 m tall (Fig. 5-51), creeping rhizomes and root penetrating to depth of about 30-50 cm (Fig. 5-52). Leaf blades are up to 40 cm long, inflorescence is an open panicle to 35 cm long (Fig. 5-52). It grows in water up to 1 m deep, but preferably 30-60 cm, on creek and river banks, and margins of dams and marshes (Májovský et al., 1982; Sainty and Jacobs, 2003). *G. maxima* grows across the whole Europe and temperate areas of Asia, North America. and Australasia. It prefers lowlands.

G. maxima has high nutrient requirements and generally forms pure stands in wetland ecosystems that are rich in nutrients and organic matter (Sundblad and Wittgren, 1989). Under such conditions, *Glyceria* can even replace *Phragmites* (Hroudová and Zákravský, 1999; Crawford and Brändle, 1996). *G. maxima* was indicated to stimulate nitrifying bacteria in its rhizosphere (Both et al., 1992; Bodelier et al., 1996). *G. maxima* is highly aerenchymatous (Smirnoff and Crawford, 1983) and the inability of its roots to survive anoxia necessitates continuous oxygenation of the root tissue resulting in oxygen leakage into rhizosphere (Brändle and Crawford, 1987; Rees et al., 1987).

HF constructed wetlands planted with *G. maxima* are in operation in the Czech Republic (Fig. 5-53), New Zealand (van Oostrom and Cooper, 1990). Ozimek and Klekot (1979) and Obarska-Pempkowiak and Ozimek (2003) reported the use *Glyceria* in FWS constructed wetlands. *G. maxima* is also a very productive species with maximum aboveground biomass values in natural stands (see Table 3-2) exceeding 2,500 g DM m^{-2}. Obarska-Pempkowiak and Ozimek (2003) reported biomass values up to 3,200 g DM m^{-2} in FWS constructed wetland at Bielkovo, Poland.

Figure 5-51. Glyceria maxima in HF CW Sedlce, Czech Republic. Photo by Jan Vymazal.

Figure 5-52. *Glyceria maxima*. Left: belowground structures, Right: inflorescence. Photos by Lenka Kröpfelová.

Figure 5-53. HF constructed wetland Sedlce, Czech Republic planted with *Glyceria maxima*. For detail see *Fig. 5-51*. Photo by Lenka Kröpfelová.

5.2.5.5 *Scirpus* spp. (Bulrush)

Species belonging to the genus *Scirpus* (Cyperaceae) are annual or perennial herbs which grow in tufts or large colonies (Fig. 5-54). Stems are sharply triangular or slightly rounded and softly angled up to 3 meters tall in some species. Leaves are reduced to bladeless sheaths in some species, blades are flat or folded from closed sheaths in others. Inflorescence (Fig. 5-55) is terminal, with spikelets in a single tight cluster or in branched arrays, subtended by one or more spreading, leaf-like bracts (in some species one erect bract appearing as an extension of the stem (Tobe et al., 1998).

Figure 5-54. Scirpus (Schoenoplectus) lacustris (top) and *Scirpus validus* (bottom) in natural stands (Paul do Boquilobo, Portugal and Perth, Australia, respectively). Photos by Jan Vymazal.

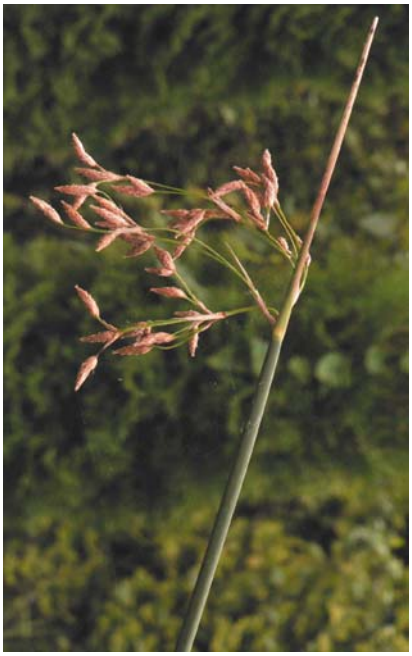



Figure 5-55. Inflorescence of *Scirpus tabernaemontani* (left) and *Scirpus lacustris* (right). Photos by Lenka Kröpfelová.

Roots penetrate up to 70-80 cm resulting in greater root-zone aeration and concomitant microbial nitrification. However, in constructed wetlands *S. validus* roots penetrate sometimes only to 10-30 cm (Tanner, 1994; Pullin and Hammer, 1991; Sievers, 1993). In mature stands growing in natural (van der Valk and Davis, 1978) and constructed wetlands (Tanner, 1994) *S. validus* shows marked season cycles of aboveground growth. Edwards et al. (1993) noted good establishment and growth of *S. validus* in constructed wetlands treating domestic wastewaters in Kentucky with relatively long periods of active growth compared to many of the other species tested.

Scirpus lacustris was used by Seidel in early stages (e.g., Seidel, 1965a, 1976) of development of constructed wetlands for wastewater treatment. However, at present, *Scirpus* is mostly used in North America, Australia and New Zealand (e.g., Tanner, 1994; Behrends et al., 1994; Wallace and Knight, 1996). The species used in HF constructed wetlands are *Scirpus (Schoenoplectus) lacustris* (L.) Palla (Bulrush), *Scirpus (Schoenoplectus) validus* (Vahl.) A. Löwe and D. Löwe (Soft stem bulrush, Lake bulrush), *Scirpus californicus* (C.A. Meyer) Steud. (Giant bulrush), *Scirpus acutus* Muhl. ex. Bigel. (Hardstem bulrush), *Scirpus cyperinus* (L.) Kunth (Wool grass) and *Scirpus (Schoenoplectus) tabernaemontani* (C.C. Gmelin) Palla (Glaucous bulrush).

Some authors classify *S. validus* as *S. tabernaemontani* (e.g., Tobe, 1998) but this classification is questionable as *S. tabernaemontani* is a haline species growing often in coastal wetlands. However, taxonomy of this genus is a bit beyond the scope of this book.

5.2.5.6 *Baumea articulata*

Baumea articulata (R.Br.) S.T. Blake (Cyperaceae) (Jointed twigrush) is a rhizomatous perennial plant up to 2.5 m tall native to Australasia and the southeast Pacific (Fig. 5-56). The only truly aquatic species of *Baumea* grows in coastal lagoons and wetlands in water up to 1 meter deep, often in deep mud where it forms large dense clumps (Sainty and Jacobs, 2003). Roots and rhizomes penetrated to depths of 30-40 cm in the gravel substratum of the constructed wetlands (Tanner, 1996). *Baumea* usually shows very slow establishment and Chambers and McComb (1994) reported poor germination from seeds, but a high rate of survival for rhizome transplants in surface-flow wetlands treating mine wastewater in western Australia. Adcock and Ganf (1994) reported a mean above and below-ground biomass of 3,800 and 3,900 g m^{-2}, respectively, for *Baumea* growing in HF constructed wetland treating tertiary sewage wastewaters in south Australia. *B. articulata* is used in HF constructed wetlands in Australia and New Zealand (Fig. 5-57).

Figure 5-56. Baumea articulata growing in a natural stand in Western Australia. Photo by Tom Headley, with permission.

Figure 5-57. *Baumea articulata* in a HF part of a hybrid constructed wetland at Otorohanga, New Zealand. Photo by Jan Vymazal.

5.2.5.7 Other plants

Cyperus papyrus

Cyperus papyrus (L.) (Cyperaceae) (Papyrus) is an erect perennial plant up to 4 meters tall (Fig. 5-58) with a short, thick, woody rhizome. It grows in slowly-flowing water to 1 meter deep. It can grow and spread vegetatively while floating. *C. papyrus* is used in constructed wetlands in Africa (e.g., Okurut et al., 1999).

Figure 5-58. Left: *Cyperus papyrus* in CW Jinja, Uganda . Photo by Frank Kansiime, with permission.

Cyperus involucratus

Cyperus involucratus Rottb. (Cyperaceae) (Umbrella sedge, syn. *Cyperus alternifolius*), a widespread horticultural species, native to Africa is a tufted perennial up to 1.5 meters tall (Fig. 5-59). It grows on creek banks and in water to 60 cm deep. Stem is triangular in cross section with leaves reduced to reddish-brown basal sheaths. Inflorescence with 12-20 leaf-like bracts each 10-30 cm long, much longer than the inflorescence (Sainty and Jacobs, 2003).

C. involucratus has been used successfully in small-scale gravel-based constructed wetlands in Australia and New Zealand, where it is a common domestic garden plant growing in both dry and wet soil conditions (Tanner, 1996). As identified by Hocking (1985), the attributes that make *Cyperus* a potentially useful plant for constructed wetlands include: year-round growth in warm temperate regions (withstanding moderate frosts), tolerance to hyper-eutrophic conditions and salinity, ease of propagation, and apparent lack of serious weed potential. Its ability to tolerate a wide range of soil moisture conditions may also be an advantage where water flows vary markedly, e.g., stormwater detention wetlands (Tanner, 1996). A low relative ability to oxygenate its root-zone may limit the potential of *C. involucratus* to enhance microbial nitrogen removal from ammonium-rich wastewaters (Tanner, 1996).

Figure 5-59. Cyperus involucratus, HF CW Waikeria, New Zealand. Photo by Jan Vymazal.

Iris spp.

Iris pseudacorus (L.) (Iridaceae) (Yellow flag) is a decorative perennial herb up to 1.5 m tall with robust rhizome (Fig. 5-60, 5-61). Stem is upright, rounded to flat and branched. It is found across the whole Europe, in the Middle East and north Africa along the ponds, lakes, slowly flowing streams and rivers and in wet meadows. *I. pseudacorus* requires permanent or temporary flooded soils rich in nutrients. In North America, *Iris versicolor* (Northern blue flag) and in Europe, *Iris sibirica* (Siberian iris), are also used for HF constructed wetlands.

Figure 5-60. Iris pseudacorus growing in a natural stand in a eutrophic pond in the Czech Republic (left) and detail of the flower (right). Photos by Lenka Kröpfelová.

Figure 5-61. HF constructed wetland at Žitenice, Czech Republic planted with *Iris pseudacorus*. Photo by Jan Vymazal.

5.2.5.8 Locally used plants

From a theoretical point of view, many emergent species could be used for HF constructed wetlands. However, in reality, only a limited number of species has been used so far. Table 5-5. presents examples of plant species reported from HF constructed wetlands in the literature (besides those listed in previous sections 5.2.5.1 to 5.2.5.7).

Table 5-5. Examples of locally used plants in HF constructed wetlands.

Scientific name	Common name	Location	Ref.
Thysanolaena maxima	Tiger grass	Mayotte near Mozambique	1
Canna sp.	Canna lily	Mayotte near Mozambique	1
Canna glauca	Canna lily	Central America	2
Pennisetum purpureum	Napier grass	Central America	2
Zizaniopsis bonariensis	Espadańa	Brazil	3
Acorus calamus	Sweet flag	Ohio, USA	4
Lobelia cardinalis	Cardinal flower	Ohio, USA	4
Asclepsis incarnata	Swamp milkweed	Ohio, USA	4
Pontederia cordata	Pickerelweed	Ohio, USA	4
Canna x. generalis	Common garden canna	Kentucky, USA	5
Hibiscus moscheutos	Hibiscus	Kentucky, USA	5
Festuca arundinacea	Fescue	Kentucky, USA	5
Hemerocallis fulva	Day lilies	Kentucky, USA	5
Mentha spicata	Spearmint	Kentucky, USA	5
Coix lacryma-jobi	Job's tears	Costa Rica	6
Colocasia esculenta	Cocoyam, Wild taro, Elephant ear	Tanzania	7
Sorghum halapense	Johnson grass	Jordan	8
Carex acutiformis	Lesser pond sedge	Denmark	9
Carex gracilis	Slender sedge	Slovenia	10
Gynerium sagittatum	Wild cane	Jamaica	11
Phylidrum lanuginosum	Frogsmouth	Australia	12
Kyllinga erecta	Greater kyllinga	Tanzania	13

1- Esser et al. (2006), 2- Platzer et al. (2002), 3- Philippi et al. (2006), 4- Steer et al. (2002), 5-Karathanasis et al. (2003), 6-Dallas et al. (2004), 7-Mbuligwe (2004), 8-Al-Omari and Fayyad (2003), 9-Brix and Schierup (1989a,b), 10-Urbanc-Berčič et al. (1998), 11-Stewart (2005), 12-Browning and Greenway (2003), 13-Haule et al. (2002)

5.2.5.9 Planting

There are several ways of planting HF constructed wetlands. The best studied and most commonly used plant, *Phragmites australis*, has four distinct kinds of material which can be used to establish vegetation of *Phragmites* in a constructed wetland: transplanted rhizomes, stem cuttings, seedlings and seeds (Brix in Vymazal et al., 1998b). Most of the following text on *Phragmites* planting is taken from Brix (in Vymazal et al., 1998b, with permission of Backhuys Publishers.)

Horizontal and vertical rhizomes with at least one shoot or bud can be planted directly in the reed bed (Fig. 5-62). The success of this technique depends on the developmental stage of the shoots and on the degree of damage to them during sampling and planting (Véber, 1978). Experience show that the survival of planted rhizomes is roughly 50% and seems to be greater at low water levels (5 cm below the surface of the bed) compared to high water levels (5 cm above the bed surface). Splitting rhizomes into smaller fragments will decrease survival. Rhizomes fail to grow under completely flooded conditions because of inadequate oxygen supply, despite the fact that the water may be saturated with oxygen. For the same reason, in flooded beds it is vital not to break or damage the rhizomes in order to maintain the aerial connection between the atmosphere and the developing buds. Rhizomes have to be planted obliquely with some part aboveground and above the water level.

Above the ground, *Phragmites* shows little tendency to spread into bare areas during the first summer, despite substantial rhizome growth below ground. This is because buds forming after the emergence period in spring, accumulate near the surface until emergence begins in the following spring. Hence the cover of shoots during the first year is largely dependent on the initial plant density.

Figure 5-62. Phragmites australis rhizome (left) and clumps of rhizomes (right) suitable for planting in constructed wetlands. Photos by Jan Vymazal.

Large clumps (cca. 20 cm x 20 cm, Fig. 5-62, see also Fig. 5-38) transplanted in spring will almost certainly establish successfully. However, the lack of a suitable supply of *Phragmites* and the high cost of excavating and transplanting clumps, make this method practical only in small beds.

Stem cuttings can be successfully planted directly into a water-saturated bed under field conditions in mid-May at a survival rate of approximately 35%. This avoids the expense of greenhouse propagation and minimizes the disturbance due to transplanting growing plants. Stem cuttings need to be at least two nodes long, but should not include the immature nodes at the base of the stem. Trimming of the upper leafy part of the stem will increase the percentage success. Later in the season, the percentage will be less. Thus stem cuttings can only be taken during May and June.

The seed production in reed stands varies considerably among sites (or clones). Some clones produce no seed, and the seed production in other clones may vary from <100 to >1,000 seeds per flower (Haslam, 1973a). The reason for this is unknown. The percentage germination of seeds may vary from 2 to 96% among panicles from a small plot of reeds (Haslam, 1973a). Fresh seeds may need to be chilled and stored at 5°C for several months to enhance germination. Seeds germinate successfully on damp soil or moist filter paper under controlled laboratory conditions, with a day/night temperature regime of 30°C/20°C (Haslam, 1973a; Cooper et al., 1996). A high and fluctuating temperature is apparently preferred and this can occur in the top soil in spring. Under these conditions seeds will germinate after 5 or 6 days, and germination will usually be completed within 7 to 9 days. The final percentage of seed germination ranges between 30 and 100%. Direct sowing of seeds into constructed wetlands is not used, however, seeds may be used to grow seedlings in the greenhouse or nursery.

In theory, there is no reason why *Phragmites* beds should not be established directly from seeds. Indeed, for very large areas it may be the only practical method. Establishment from seeds should be almost as rapid as from seedlings or rhizomes. However, to our knowledge no practical experience is as yet available on this technique.

Germination and survival of *Phragmites* seedlings in natural habitats is poor (Haslam, 1971b, 1973a). Even if the seeds germinate successfully, the young seedlings are vulnerable to weed competition, lack of light, low nutrient, frost, drought and flooding. However, under greenhouse conditions all these factors can be controlled and seedling mortality is negligible. Seedlings can be produced by germination in a greenhouse during winter, by scattering pieces of fertile flower heads onto the surface of moist seedling compost and pressing the material into firm contact with the soil.

Seedlings doubled their shoot weight every 5 to 6 days, but despite this, it will take about two months before the seedlings have reached a height of 20 cm (Haslam, 1971b). Seedlings should be transplanted into individual 5-cm pots after 40-50 days, when they are approximately 10 cm high. After about three months, by which time the seedlings have produced some rhizome growth, they can be transferred to 7-cm plastic pots for further propagation. Seedlings planted in May generally show a 100% survival by September. Although young seedlings cannot survive complete flooding (Weisner et al., 1993), once they have reached a height of 20 cm they will grow best when the water table is 5 cm above the ground level.

Presently, the use of potted seedlings is the most commonly used technique in northern Europe. A density of 4 plants per square meter is generally used. However, it has been found that seedlings could successfully be grown outdoors as well. Also, it has been found that planting of bare-root seedlings is possible and advantageous as compared to potted seedlings. The major advantage is that large number of bare-root seedlings could be easily transported. Also, the soil from pots may be washed out and contribute to bed clogging.

Phragmites growth is quite slow as compared to other plants used in constructed wetlands and it has been found that it usually reaches the maximum aboveground biomass in 3 to 4 years (Vymazal and Kröpfelová, 2005). Also, it takes at least one full growing season to provide a dense cover (Fig. 5-63). It has also been reported that *Phragmites* undergoes the process of "self-thining" (Mook and van der Toorn, 1982; Vymazal and Kröpfelová, 2005), i.e. the number of stems decreases over time while the weight of individual shoots increases.

Figure 5-63. Phragmites australis: 10 months after planting in May 2001 (left) and 14 months after planting in September 2001 (right) in HF CW Mořina, Czech Republic. *Phragmites* band is in between two bands of *Phalaris arundinacea*. Photos by Jan Vymazal.

Figure 5-64. Bare-root seedlings of *Phalaris arundinacea* (left) and planting into a constructed wetland Libníč, Czech Republic. Photos by Lenka Kröpfelová.

Phalaris arundinacea growth is much faster than that of *Phragmites australis* – *Phalaris* reaches its maximum aboveground biomass during the second growing season and during the first growing season already forms a very dense cover (Vymazal and Kröpfelová, 2005, see also Fig. 5-63). *Phalaris* is commonly planted in the form of bare-root seedlings which are trimmed about 30 above the ground and planted in hand-dug holes (Fig. 5-64).

Phalaris increases both its above- and belowground biomass shortly after planting (Fig. 5-65). This is very important in temperate and cold climates as insulation of the surface is very important.

Figure 5-65. Phalaris arundinacea: 6 weeks (top) and 4 months (bottom) after planting. Photos by Lenka Kröpfelová.

Glyceria maxima is best planted in the form of grown seedlings. It creates intensive belowground biomass immediately after planting while aboveground biomass substantially increases during the second growing season (Fig. 5-66). Both *Phalaris* and *Glyceria* grow best after planting when shallowly flooded for about 2 months.

Figure 5-66. Glyceria maxima – constructed wetland Sedlce, Czech Republic. Top: two months after planting, September 2005, middle: May 2006, bottom: August 2006. Photos by Lenka Kröpfelová.

5.2.5.10 Harvesting

The necessity of harvesting of vegetation in constructed wetlands with horizontal sub-surface flow is still not unanimous. However, it seems that harvesting does not substantially influence the treatment process. In addition, reliable information on the harvesting effect is missing. Therefore, it is necessary to use the information from natural stands.

Güsewell et al. (2000) reported that various mowing regimes of *Phragmites australis* is Swiss fens had no effect on shoot number and size. Mowing in winter every three years reduced shoot size in the year after mowing, but not on the long term. Mowing every three years in late summer reduced the shoot size compared to unmown plots on the short term, but this effects nearly disappeared on the long term, after mowing had become biennial. Granéli (1989) reported that removal of standing litter in Lake Tåkern, Sweden, in winter caused a doubling or reed aboveground biomass at the end of the following growing season. The positive effect on aboveground biomass was mainly caused by increased shoot density, while shoot height and mean shoot weight were less affected by litter removal. Standing, dead shoots reduce the light available for the new shoot generation. The author also suggested that this shading effect may cause the invasion of other species into the reed stand. Also, the increase shoot production after removal of standing litter may be the result of elimination of insect damage (van Toorn and Mook, 1982; Granéli, 1989). There are many types of insects attacking reed (Durska, 1970) and some of them are dependent on old shoots for over-wintering (van Toorn and Mook, 1982).

If standing litter of *Phragmites australis* is removed in winter, frost can kill the young shoots (Haslam, 1969). A replacement crop of new side shoots is then produced. The second generation is composed of tiny shoots and aboveground biomass is lowered in comparison to uncut stands.

Rolletschek et al. (2000) reported that the elimination of old culms through mowing resulted in a lower ventilation efficiency due to a high counterpressure of rhizomes. The corresponding gas flow rates were reduced to 38% of the value in control stands, indicating a strongly impaired oxygen supply to basal and belowground plant parts after mowing. If not harvested, dead culms of *Phragmites australis* can remain standing for two or more years (Roman and Daiber, 1984; Meyerson et al., 2000, see also Fig. 5-67).

It has been shown that *P. arundinacea* is able to tolerate three cuttings a year with slightly increasing or equal biomass production and increase in nutrient uptake (Lawrence and Ashford, 1969; Horrock and Washko, 1971; Marten and Hovin, 1980; Kröpfelová and Vymazal, 2006). At the end of the growing season standing litter bends down and creates a compact layer which provides very good insulation of the bed (Fig. 5-67). On the other hand, if not harvested the litter is usually invaded by weeds such as *Urtica dioica* (Stinging nettle, see also section 5.2.5.11).

Removal of nutrients via harvesting in temperate and cold climates is usually negligible as compared to inflow loading in constructed wetlands treating wastewaters (e.g., Vymazal, 2004a,b, 2005a, 2007, see also sections 5.4.3 and 5.4.4). The situation may be different in the subtropics and tropics with growing season throughout the whole year where plants can be harvested several times during the year.

Figure 5-67. *Phalaris arundinacea* (in front) and *Phragmites australis* (in the back) standing litter in CW Mořina, Czech Republic, in January 2007. Photo by Jan Vymazal.

If harvested, plants are usually removed at the end of the winter season when the danger of heavy frosts is not high. Plants could also be cut down in late autumn and left on the surface to provide winter insulation. Harvested material may be composted but most frequently is it burned outside the beds.

5.2.5.11 Weeds

In constructed wetlands, "weeds" may be seen as plant species, which were not intentionally planted in the system (Table 5-6). However, it is question if the presence of "weedy" species causes any problems to the system performance. Unfortunately, there are no studies which would monitor the performance of one constructed wetland with and without the weedy species at the same time.

However, plant species which are commonly used in HF constructed wetlands are usually very aggressive and mostly possess the weed properties. Therefore, if properly planted, intentionally planted species do not allow other species to invade the vegetated beds. The presence of “weeds” is usually limited to bed margins. Also, weedy terrestrial species could be effectively eliminated by shallow flooding of the bed after the plantation.

The presence of weeds may be regarded as undesirable because they detract from the appearance of the system. Also, presence of woody species such as willows, alders or poplars may cause problems as the roots of these species may penetrate the liner of the system.

Phalaris arundinacea stands, especially when the biomass is not harvested, is very often invaded by *Urtica dioica* (Stinging nettle). *U. dioica* very often occurs with *P. arundinacea* in wet habitats (Kopecký and Hejný, 1965; Menges and Waller, 1983; Šrůtek, 1993; Klimešová, 1994; Klimešová and Čížková, 1996) where it occupies elevated places. In constructed wetlands the rhizome system of *U. dioica* develops in the layer of *Phalaris* litter and does not penetrate to the saturated substrate.

Weed species may indicate the hydrologic status of the bed. In case of hydraulic underloading typical terrestrial species may occur while in case of permanent hydraulic overloading typical aquatic plants may occur.

Table 5-6. Examples of “weed” species growing in HF constructed wetlands.

Latin name	**Common name(s)**
Anagalis arvensis	Scarlet pimpernel
Atriplex patula	Fat-hen saltweed
Epilobium hirsutum	Hairy willow-herb
Holcus lanatus	Velvet grass
Impatiens capensis	Jewelweed
Lythrum salicaria	Purple loosestrife
Polygonum persicaria	Lady’s thumb smartweed
Populus deltoides	Cottonwood
Rumex acetosa	Green sorrel
Rumex obtusifolius	Bitter dock, Broad-leaved dock
Urtica dioica	Stinging nettle
Salix spp.	Willows

5.2.6 Sizing the bed

The size of wetland beds has been designed according to many models beginning with simple “rule of thumb” and complex dynamic, compartmental models on the other hand of the design spectrum. In between these two solutions, plug-flow first order k (or k-C*) models (e.g. Kadlec and Knight, 1996) and Monod-type equations (e.g., Kemp and George, 1997; Mitchell and McNevin, 2001) have been used (for summary see Rousseau et al., 2004a).

The initial “rule of thumb” design “models” usually used hydraulic and organic loading rates (HLR and OLR, respectively). For example, Wood (1995) suggested HLR between 0.2 and 3.0 cm d^{-1} and OLR < 75 kg BOD_5 ha^{-1} d^{-1} and Watson et al. (1989) suggested HLR between 3.9 and 4.7 cm d^{-1}.

A large number of wetland systems have shown an exponential decrease of pollutant concentration level with the distance through the wetland from inlet to outlet. The observation is consistent with a first-order removal model, with the removal rate being proportional to the pollutant concentration (Brix, 1998). The removal can thus empirically be described with first-order plug-flow kinetics (Kickuth, 1981; Reed et al., 1988; Watson et al., 1989; Cooper, 1990; Wood, 1994):

$$C_o/C_i = \exp\,[-k_V t] \tag{5.2}$$

where C_o = ouflow concentration (mg l^{-1})
C_i = inflow concentration (mg l^{-1})
k_V = temperature-dependent, first-order volumetric reaction rate constant (d^{-1})
t = hydraulic residence time (d)

Hydraulic residence time t can be described as

$$t = V\,n\,/Q = A\,d\,n\,/\,Q \tag{5.3}$$

where V = volume of the bed (m^3)
n = porosity of the substrate as a decimal fraction (ratio between void spaces in the bed and total volume of the bed
Q = average flow rate (m^3 d^{-1}) (for impact of precipitation and evapotranspiration see Kadlec (1997)
A = area of the bed (m^2)
d = saturated bed depth (m)

Combination of Equations (5.2) and (5.3) yields

$$A = Q\,(\ln C_i - \ln C_o)/\,k_V\,d\,n \tag{5.4}$$

with

$$k_V\,d\,n = k_A \tag{5.5}$$

where k_A = first-order areal rate constant (m d^{-1})

Combination of Equations (5.4) and (5.5) yields

$$A = Q\,(\ln C_i - \ln C_o)/\,k_A \quad (5.6)$$

or for the k-C* model (Kadlec and Knight 1996, see section 4.1.4.1 for more details)

$$A = Q \ln\,[(C_i - C^*)/C_o - C^*)]\,/\,k_A \quad (5.7)$$

The temperature dependence of the rate constant is commonly described by Arrhenius equation:

$$k_{A,T} = k_{20}\Theta^{T-20} \quad (5.8)$$

where Θ is temperature coefficient for rate constant. For example, Kadlec and Knight (1996) suggested that removal of BOD, TSS, P and fecal coliforms in HF constructed wetlands is generally found to be independent of temperature (Θ = 1.00) whereas ammonia-N (Θ = 1.04), nitrate-N (Θ = 1.09) and total N (Θ = 1.05) removal is negatively influenced by lower temperature.

Reed et al. (1988) pointed out that the rate constant k_{20} for a particular system may be related to the porosity of the medium used to construct the bed:

$$k_{20} = k_o(37.31n^{4.172}) \quad (5.9)$$

where k_{20} = design rate constant at 20°C for the selected bed medium (d^{-1})
k_o = the "optimum" rate constant for a medium with a fully developed root zone (d^{-1})
= 1.839 d^{-1} for typical municipal wastewaters
= 0.198 d^{-1} for industrial wastewaters with high COD
n = total porosity of the medium (decimal fraction)

Rousseau et al. (2004a) pointed out that calibration of the parameters k, C* and Θ is mostly done on the basis of inflow/outflow concentrations, and not on the basis of transect data, although the latter are to be preferred for calibration purposes (Kadlec, 2000). Because these parameters lump a large number of other characteristics representing the complex web of interactions in a constructed wetlands as well as external influences such as weather conditions, a large variability can be observed in reported k_A, k_V, C* and Θ values (Rousseau et al., 2004a).

Marsili-Libelli and Checchi (2005) pointed out that recent studies suggest the the interdependence between hydraulics and kinetics is so strong and influenced by such a large number factors (Kadlec, 2000) that even the last generation of models (Wynn and Liehr, 2001; Mashauri and Kayombo, 2002; Langergraber, 2003; Kincanon and McAnally, 2004) is inadequate in fully explaining the observed behaviors and providing a reliable extimation of their parameters.

As BOD was the primary target, the value of first order areal rate constant k_A used for the design was k_{BOD}. Formerly proposed value of 0.19 m d^{-1} by Kickuth (1977, 1981) resulted in too small area of the bed and consequently lower treatment effect. The first field measurements showed that the value of k_{BOD} is usually lower. Schierup et al. (1990b) reported a value of 0.083 ±0.017 m d^{-1} for 49 systems in Denmark and Cooper (1990) reported values of 0.067 to 0.1 m d^{-1} in the United Kingdom. In the Czech Republic (Vymazal and Kröpfelová, unpubl.), the average value for 31 constructed wetlands was 0.087 m d^{-1} but the values varied widely (± 1.2 m d^{-1}). Kadlec and Knight (1996) suggested k_{BOD} = 0.49 m d^{-1}, however, this value probably vastly underestimates the constructed bed area.

Brix (1998) pointed out that in theory, the rate constant should be a constant, and therefore ideally independent of inlet concentration and loading rate. However, it has been found that rate constants generally increase with hydraulic loading rate and BOD_5 mass loading rate (Fig. 5-68).

In addition, in systems which have been in operation for at least ten years a steady increase in k_{BOD} value was observed (Fig. 5-68). The average k_{BOD} value for 66 village systems after two years of operation was 0.118 ± 0.022 m d^{-1} (Brix, 1998). At present, using a large set of field measurements from many operational systems, the value of 0.1 m d^{-1} (Cooper et al., 1996) is considered as appropriate in order to provide sufficient removal of organics and suspended solids. This generally means that the value of A_h is about 5 m^2 (see also Table 5-7).

However, HF constructed wetlands may be designed with other target parameters than BOD_5. Based on North American experience Kadlec and Knight (1996) developed following values of areal rate constant k_A adjusted to 20 °C for HF constructed wetlands: TSS: 2.74 m d^{-1}, total-N: 0.074 m d^{-1}, organic-N: 0.096 m d^{-1}, ammonium-N: 0.093 m d^{-1}, nitrate-N: 0.137 m d^{-1} and total-P: 0.0329 m d^{-1}. Brix (1998) reported k_A-values based on the Danish experience as follows: total-N: 0.0329 m d^{-1} and total-P: 0.0247 m d^{-1}. Kröpfelová and Vymazal (unpubl.) found average k_A values for the Czech HF systems: TSS: 0.085 m d^{-1}, total-P: 0.026 m d^{-1}, total-N: 0.025 m d^{-1}, ammonium-N: 0.024 m d^{-1} and nitrate-N: 0.039 m d^{-1}. Those numbers are much closer to those reported by Brix (1998) from Denmark as compared to figures reported by Kadlec and Knight (1996). However, it is important to realize that k_A value itself does not say anything about the treatment performance of the system. When a constructed wetland is designed with more target parameters at the same time, e.g., BOD_5, TSS, nitrogen, phosphorus, then it is necessary to select the largest area according to all four calculations.

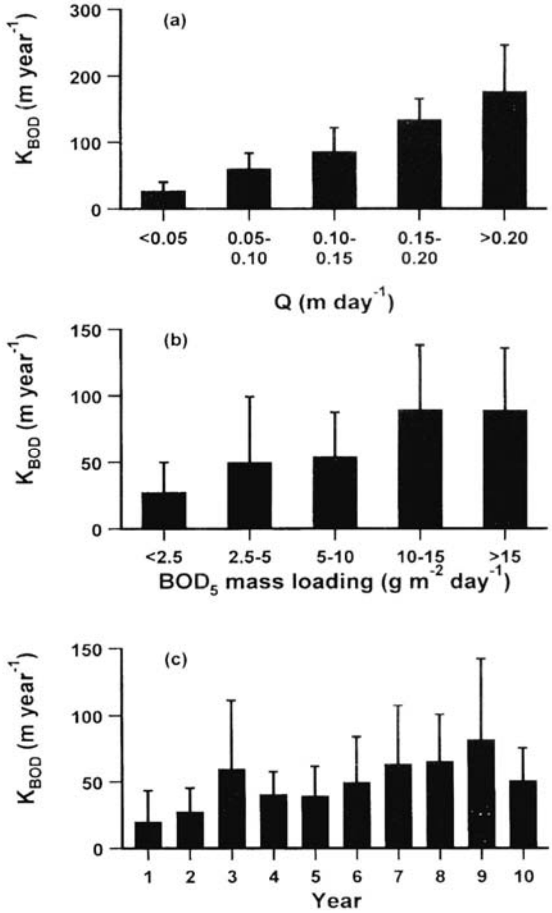


Figure 5-68. Calculated rate constants (mean ± 1SD) for the degradation of BOD_5 (k_{BOD}, m yr^{-1}) in 66 Danish constructed wetlands as a function of (a) mean hydraulic loading rate (Q, m d^{-1}) and (b) BOD_5 mass loading rate (g BOD_5 m^{-2} d^{-1}). (c) Average k_{BOD} (± 1SD) for 13 systems that have been in operation for at least 10 years. From Brix (1998) with permission from Backhuys Publishers.

The mechanistic models (e.g. Wynn and Liehr, 2001; Langergraber, 2003) on the other hand did not offer too much help in designing due to many assumptions and empirical relations that are not physically based and thus corrupt model output. However, this model is a useful tool to gain understanding of certain processes and it is well able to demonstrate several interactions within the wetland system (Rousseau et al., 2004a).

At present, the state of the art k-C* model seems to be best available design tool if the designer makes sure that all the assumptions are fulfilled and if he(she) is aware of many pitfalls in the model. Concerning the issue of parameter uncertainity, it is advisable to implicitly take this into account during the design. If possible, parameter values such as climatic conditions, wastewater composition, filtration material or macrophyte species, should be used from constructed wetlands with similar operational conditions as the one to be constructed (Rousseau et al., 2004a). Another uncertainty comes from the fact that background concentrations inevitably change over the operation period as the system mature. Also, Stein et al. (2006) suggested that C* is temperature-dependent. Kadlec and Knight (1996) suggested C* for $BOD_5 = 3.5 + 0.053C_{in}$. However, Cooper and Green (1998) reported at least 15 HF tertiary systems in United Kingdom which provided outflow BOD_5 concentration less than 2 mg l^{-1}.

Table 5-7. Examples of design criteria for surface area of HF constructed wetlands. In most countries, population equivalent (PE) equals the number of connected people for the design. However, in reality, 1 PE (i.e., 60 g BOD_5 d^{-1} per person) does not equal 1 connected person in terms of BOD_5 for small systems. In these systems, 1 PE could for various reasons equal up to 2 connected people. For more see section 5.4.1. The criteria from the United States and Spain are based on organic loading and recalculated to PE.

	Area (m^2 PE^{-1})	**Comments**	**Reference**
Austria	6	For BOD removal, maximum 11.2 g BOD_5 m^{-2} d^{-1}	ÖNORM 2005 (2005)
Belgium	>3	For Flemish part only	VMM (2002)
Czech Republic	5	For BOD and TSS removal; maximum 10 g BOD_5 m^{-2} d^{-1}	ČSN 75 6402 (1998)
Denmark	5	Minimum area 25 m^2	Ministry of the Environment (1999)
Germany	5	Minimum area 20 m^2	ATV-A 262 (1998)
Italy	4-6	< 2000 PE, appropriate treatment	Pucci et al. (2004)
Spain	> 10*	*For inflow BOD_5 < 250 mg l^{-1}, load should be < 6 g BOD_5 m^{-2} d^{-1}	García and Corso (2007) García et al. (2004c)
United Kingdom	5	Secondary treatment	Cooper et al. (1996)
	0.5-1	Tertiary treatment	
U.S.A.	10* (38)**	*6 g BOD_5 m^{-2} d^{-1} for outflow BOD_5 30 mg l^{-1},**1.6 g BOD_5 $m^{-2}d^{-1}$ for outflow BOD_5 20 mg l^{-1}	US EPA (2000)
U.S.A.	7.5* (15)**	*8 g BOD_5 m^{-2} d^{-1} for outflow BOD_5 30 mg l^{-1}, **4 g BOD_5 m^{-2} d^{-1} for outflow BOD_5 25 mg l^{-1} (50% of the time)	Wallace and Knight (2006)

Rousseau et al. (2004a) in their comprehensive analysis of HF CWs models pointed out that different models of the HF constructed wetlands and the numerous different parameter values obviously raise a question which one should be used and which one is the most reliable one. It has been shown that predicted required surface areas are highly variable among models but also within the same model category. The authors further concluded that the rules of thumb seem to be the more conservative design models (Table 5-7). Since these models are easily applicable, designers could be tempted to stick with those models. However, they may guarantee good quality effluent but they will likely be counteracted by economic constraints as conservative design tend to increase the investment costs.

5.2.6.1 Bed width

The width of the bed which is necessary to keep the sub-surface flow has been determined using Darcy's Law (Eq. 5.10) and effective water depth in the bed (e.g., U.S. EPA, 1988; Watson et al., 1989; Cooper, 1990; Hu et al. 1994; Cooper and Green, 1998):

$$Q = k_f A_c (dH/ds) \tag{5.10}$$

where Q = average flow ($m^3 d^{-1}$)
k_f = hydraulic conductivity for the fully developed bed ($m d^{-1}$)
A_c = cross-sectional area (m^2)
dH/ds = bed bottom slope or hydraulic gradient ($m m^{-1}$)

Wallace and Knight (2006) pointed out that although recommended in some design methods (e.g., Cooper, 1990), the bed bottom slope does not drive the flow and cannot be used to as a substitute for the actual hydraulic gradient. Practical designs use a flat (or nearly flat) bed with a very conservative (large) cross-sectional area. This approach reduces the impact of flow on the water surface profile, since the head loss across the bed is low. The range of water level adjustment at the outlet can also easily accommodate the range of flow conditions that will be encountered during the start-up, low flows and peak flows.

Wallace and Knight (2006) also reported that for the flat-bed approach to work, the cross-sectional area of the bed must be designed conservatively to avoid surfacing (ponding) problems at the inlet end of the bed. This is generally done by one of three ways:

1. By applying a large safely factor to the clean bed hydraulic conductivity.
2. By assuming a conservative hydraulic conductivity value.
3. By limiting the organic or TSS loading on the cross-sectional area.

Early design references suggested designing the bed based on actual clean bed hydraulic conductivity (e.g., U.S. EPA, 1988; Reed et al., 1995). Later references recognized then potential for bed clogging and suggested that the clean bed hydraulic conductivity be reduced by a factor 10 (Kadlec and Knight, 1996) or even a factor 100 (U.S. EPA, 2000). Hydraulic conductivity (k_f) can be estimated according to the Hazen's formula (Bahlo and Wach, 1993; Brix et al., 1993; Blažejewski et al., 1994; see section 5.2.3.2 for description of d_{10}):

$$k_f = 0.01\ d_{10}^2 \tag{5.11}$$

Wallace and Knight (2006) pointed out that this relationship is suggested under the conditions that the bed material contains less than 5% fines, have a uniformity coefficient UC (d_{60}/d_{10}) < 5, and that the d_{10} of the medium be greater than 0.2 mm. Since the hydraulic conductivity should be less than 0.1 m s^{-1}, this relationship appears to be conservative only for fine bed media ($d_{10} \leq 3$ mm). German guidelines (ATV 262, 1998) recommend reducing the calculated hydraulic conductivity by a safety factor of 10 for design purposes. More rigorous methods to calculate hydraulic conductivities have been presented by Kadlec and Knight (1996).

The width of the bed derived from Equation (5.10) may be too long and therefore the inlet zone is usually divided into several parallel beds. This enables better wastewater distribution anlong the whole inlet length and it also allows for more operation and maintenance flexibility.

5.2.6.2 Bed configuration

The early HF constructed wetlands were usually composed of one bed irrespective of the size and therefore, some of the early beds were extremely large (Fig. 5-69). This caused problems with water distribution across the inlet zone, short circuiting and also with bed maintenance. The problems with water distribution across the inlet zone were solved by subdividing the inlet zone into two or more separate units that could be loaded separately in order to get better control of the distribution of wastewater (Brix, 1998).

Figure 5-69. HF constructed wetland at Freethorpe, United Kingdom with one bed of 3,500 m^2. Photo by Jan Vymazal.

At present, onsite systems still use one bed. However, for larger systems (cca. > 50 PE) many variations exist with the most common ones presented in Figure 5-70. These configurations enable better water distribution and maintenance of the system. The number of beds is variable, for example in constructed wetland at Beja, Portugal, the total area of 22,800 m^2 is divided into 32 beds.

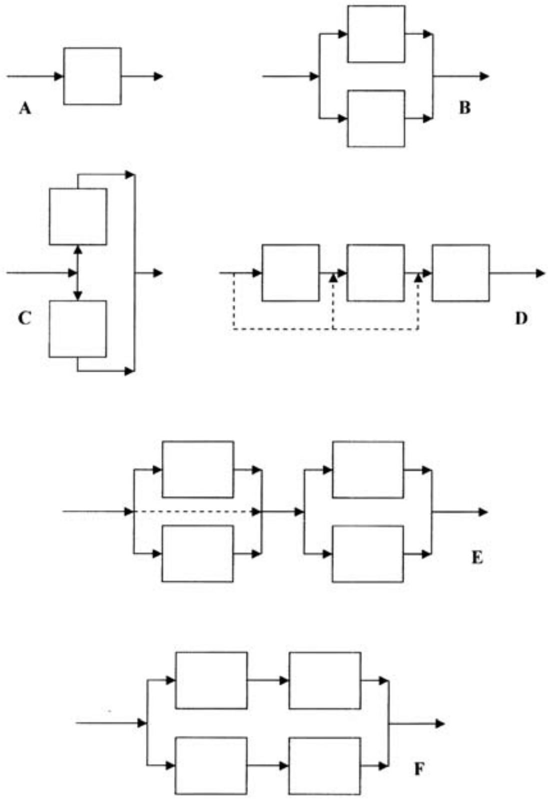


Figure 5-70. Various configurations of beds in HF constructed wetlands. A-single bed used for small systems, B, C- two parallel cells, D-beds in series with optional step-loading or bypass, E-two parellel cells in series, F-two parallel series of cells. Based on Vymazal (1998) with permission from Backhuys Publishers.

5.3 Investment and O & M costs

5.3.1 Capital costs

Investment costs of horizontal flow constructed wetlands are summarized in Table 5-8. It is quite obvious that costs vary widely among countries. Also, due to inflation, it is difficult to compare current prices with the older ones. For example, the average investment cost for four systems built during the period 2003-2006 in the Czech Republic (including pretreatment) was € 157 m^{-2} and € 631 PE^{-1}. However, analysis of the construction costs of 12 HF systems built in the period 1992-1996 revealed that the respective amounts were only € 57 m^{-2} and € 266 PE^{-1} (Cost are recalculated using the current EUR/CZK ratio of 1:28). Also, the construction costs decrease with the size of the beds (Fig. 5-71). Silva and Braga (2006b) reported a good correlation between the size of HF constructed wetlands and the investment costs for systems up to about 1,000 PE:

$$\text{Costs } (€\ PE^{-1}) = -297 \ln PE + 2103 \quad (r^2 = 0.58) \tag{5.12}$$

Table 5-8. Capital (investment) costs for HF constructed wetlands treating mostly municipal or domestic wastewater.

Reference	**Country**	**Cost per m^2**	**Cost per PE**	**No. of systems**
Stewart (2005)	Jamaica	$ 86 (74-97)		2
Dzikiewicz (1996)	Poland	€ 31 (10-83)	€ 121 (25-411)	19
Rousseau et al. (2004b)	Belgium	€ 257 (237-277)	€ 1,258	2
IRIDRA (2002)	Italy	€ 125 (38 – 247)	€ 441 (80-1,240)	33
Masi et al. (2006)	Italy	€ 115 (101-129)	€ 377 (362-392)	2
Steiner and Combs (1993)	USA	$ 74 (27-144)		5
Haberl et al. (1998)	Austria		€ 1,000	NS
Puigagut et al. (2007)	Spain		€ 503 (107-1553)	8
Billore et al. (1999)	India	$ 29		1
Platzer et al. (2002)	Central America	$ 61 (22-229)	$ 79 (34-103)	10
US E.P.A. (2000)	USA	$ 67 (32-125)		5
Dallas et al. (2004)	Costa Rica	$ 33		1
Vymazal and Kröpfelová (unpubl.)*	Czech Republic	€ 157 (100-228)	€ 631(552-750)	4
De Morais et al. (2003)	Portugal	€ 96		**
Nogueira et al. (2006)	Spain		€354 (195-645)	5***
Seyring and Kuschk (2005)	Germany		€150-1,500	****
Seyring and Kuschk (2005)	Mexico		€150-400	****
Shrestha et al. (2001a)	Nepal	$ 31-72		NS

*systems built in 2003-2006, **estimate for a system for 1,000 PE (without pretreatment), ***Canary Islands and Andalusia, ****less than 2000 PE, NS=not specified

Limited information exists on the detailed investment costs evaluation. However, available data indicate that the most expensive part of the investment cost is filtration bed (Table 5-9) and especially the filtration media (Table 5-10) which comprise 27-53% of the filtration bed costs. Also, Dallas et al. (2004) reported the single most expensive component of the HF constructed wetland in Costa Rica was the crushed rock which represented approximately 50% of the total costs.

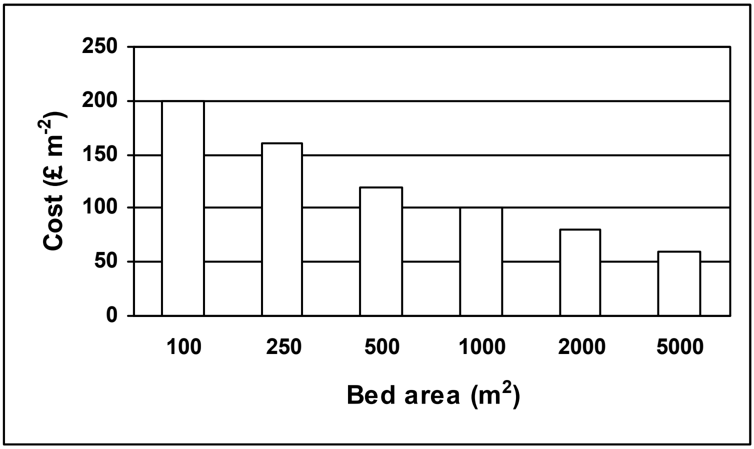


Figure 5-71. Cost of reed bed built by ARM Ltd. Data courtesy David Cooper, ARM Ltd., Rugeley, UK.

Table 5-9. Relative costs (%) of the HF constructed wetland components.

Country	**Pretreatment**	**Filtration bed**	**Miscellaneous**	**Reference**
Austria	28	51	21	Haberl et al. (1998)
Czech Republic	25	60	15	Unpubl. results
Portugal	20	60	20	Silva and Braga (2006b)

Table 5-10. Capital costs (in %) of the filtration beds of HF constructed wetlands.

	Excavation	**Gravel**	**Liner**	**Plants**	**Plumbing**	**Control structures**	**Other**
USA[1]	10.7	42.6	15.8	10.9	6.1	5.7	8.2
USA[2]	7.7	53.2	22.2	4.1	6.0	3.1	3.7
Czech Rep.	7	53	13	7	12		8
Portugal[3]	27.4	37.9	15.9	12	5.0		1.8
Spain[4]	15	27	33	2	6	5	12
Portugal[5]	12.5	37.5	25	5	11		9

[1]U.S. EPA (2000), estimate based on 4,645 m^2 system
[2]U.S. EPA (2000), average based on the minimum and maximum costs for 0.4 ha wetland
[3]de Morais et al. (2003), based on 1,000 PE system;
[4]Nogueira et al. (2006)
[5]Silva and Braga (2006b)

5.3.2 Operation and maintenance

Rousseau et al. (2005) pointed out that operational maintenance is an important factor in ensuring the longevity of HF constructed wetlands. However, it is not the frequency with which the maintenance activities are being undertaken that is having an effect on the performance of a HF system but the thoroughness with which these tasks are being carried out. This concurs with the conclusions of, for example, Cooper et al. (1996), Billeter et al. (1998), that natural treatment systems are frequently considered to be a "built-and-forget" solution and thus do not need any attention.

Cooper et al. (2005) and Rousseau et al. (2005) concluded that the most important maintenance tasks could be summarized as follows:

- Proper check of pre-treatment including regular cleaning of screens and emptying septic or Imhoff tank and grit chambers;
- regular maintenance and check of inflow distribution system;
- regular control of water level adjustment

Operational problems are mainly related to clogging phenomena, a problem commonly acknowledged among researchers (Platzer and Mauch, 1997; Blažejewski and Murat-Blažejewska, 1997; Langergraber et al., 2003).

Cooper et al. (2005) evaluated thoroughly the factors affecting the longevity of tertiary HF constructed wetlands for sewage effluents in UK. During the period November 2002 to June 2004, 126 sites were visited. The authors identified five major problems occurred either singly or in combination, namely:

1) sludge deposition on the bed surface,
2) above surface flooding (partially caused by 1), 3) and 4),
3) inlet flow distribution problems/clogging,
4) outlet collectors problems(incorrect level maintenance,
5) weed infestation.

Deposits of sludge and dead vegetation of varying depths were found on all reed beds. However, dead vegetation on the surface may not be perceived as operation/maintenance problem as it depends on the harvesting practice. Sludge deposition on the bed surface is connected with flooding. In systems where water level is kept under the surface accumulation of sludge should not occur.

Occurrence of weeds in HF constructed wetlands has been discussed in section 5.2.5.11. The weeds themselves do not influence treatment performance but they can indicate water level in the bed. Also, mixture of weeds make the wetland look bad what can raise the opinion that the system does not work properly. Cooper et al. (2005a) pointed out that weed infestation is known to be often associated with water levels in the bed that are significantly lower than the 25-30 mm below the surface used a standard. The authors recommended that if the weed infestation covered more than 25% of the bed are or if the weeds were interwined with healthy plants then these weeds should be removed by hand pulling for shallow rooted species such as willowherb (*Epilobium hirsutum*). Deeper rooted plants such as saplings of willow and alder should be cut off and then killed using a stump killer. If the weed infestation covered more than 25%, then the vegetation, sludge and top 30 cm of gravel should be removed and new gravel should be replanted. The authors mentioned that gravel cleaning may in future be done on site to allow for its recycling and obviate the needs for new gravel.

The authors suggested that the beds should be inspected at least once per month and more frequently if this is a known problem. The regular inspection should include check and if necessary reseting of the outflow level control structure. Also plants which grow into the distribution zone should be removed periodically (Cooper et al., 2005a).

Brix (1999) compared the energy requirement of constructed wetlands and conventional systems (data from U.S. EPA, 1996). Both FWS and HF CWs require less than 0.1 kWh m^{-3} as compared to extended aeration at 151 m^3 d^{-1} (2.39 kWh m^{-3}), sequencing batch reactor at 303 m^3 d^{-1} (1.13 kWh m^{-3}), extended aeration at 3,785 m^3 d^{-1} (1.06 kWh m^{-3}) and Carrousel oxidation ditch at 3,785 m^3 d^{-1} (0.51 kWh m^{-3}).

Luederitz et al. (2001) compared total material and energy requirements for constructed wetlands, technical systems and discharge to a central treatment plant 20 km away. They reported that for their function constructed wetlands needed 83% lesser energy than the central technical system and 72% lesser energy than the discharge to a central treatment plant 20 km away. For the necessary material input, the corresponding values were 76% and 63%. On the other hand, the evaluation did not differ markedly relating to their construction, because of the large amount of materials needed for wetland construction or the sewerage. Nevertheless, in case of energy, the advantage of the constructed wetlands in operation is so dominant that they overcome the small advantage of the discharge variant (in construction) in only a single year. The authors concluded that constructed wetlands are, therefore, very sustainable in rural areas and also in smaller towns (Reckerzügl and Bringezu, 1998).

The estimates of O & M costs for HF systems vary widely in the literature. U.S. EPA (2000) reports the values of 2,510 and 4,045 USD ha^{-1} yr^{-1} for systems in Carville, Louisiana and Ten Stones, Vermont, respectively. de Morais et al. (2003) estimated the O & M cost in the range of 2,000 – 6,000 EUR ha^{-1} yr^{-1} in Portugal.

Haberl et al. (1998) estimated the O & M costs in Austria of 300 EUR PE^{-1} yr^{-1} (<50 PE) and 60 EUR PE^{-1} yr^{-1} (50-500 PE). Puigagut et al. (2007) reported the O & M costs of 58 EUR (5-183) PE^{-1} yr^{-1} for 8 systems in Spain. Nogueira et al. (2006) reported operation costs (without Imhof/septic tank cleaning and sludge deposition) of EUR 42 (26-71) PE^{-1} yr^{-1} in Canary Islands and Andalusia. Masi et al. (2006a) reported that the operational costs for two HF constructed wetlands treating wastewaters from tourist facilities in Italy were 7.3 and 7.7 EUR PE^{-1} per year. Turon (2006) in her dissertation study developed a detailed protocols for assessment of operation and maintenance of HF constructed wetlands in Spain.

5.4 Treatment efficiency

In this section, we try to evaluate treatment efficiency of HF constructed wetlands based on literature survey. In the survey, only long-term results from full-scale and experimental outdoor systems have been included. Whenever possible, we use annual average values; in some cases only longer periods were available while some results are based on shorter periods but usually not shorter than several months. Only systems where both inflow and outflow data were available have been included. We have not included results from laboratory experiments and results obtained from experiments with artificial wastewaters or various growth media because this kind of experiments may provide misleading information. In Table 5-11, number of entries (annual means) for each parameter in the database is given, Table 5-12 indicates countries from which the results were included in the database and Table 5-13 gives the number of entries for major types of wastewaters.

Table 5-11. Number of entries (mostly annual means) in the database which was used for treatment efficiency evaluation. FC = fecal coliforms, TC = total coliforms, FS = fecal streptococci.

Parameter	Number of entries	Number of countries
BOD_5	1,438	40
COD	922	37
TSS	1,379	36
TP	853	36
TN	511	31
NH_4-N	1,138	37
NO_x-N	644	33
$N_{org.}$	206	17
TKN	327	24
FC	141	15
TC	75	13
FS	36	8
E. coli	48	8

When evaluating treatment performance of constructed wetlands it is important to bear in mind that secondary systems always include pretreatment stage. Unfortunately, the terminology is a bit confusing – *constructed wetland* (technology) consists of preliminary and pretreatment stages and a *constructed wetland* (vegetated bed). In the literature, mostly results from the vegetated bed stage are reported, i.e. results deal only with one part of the technology. Therefore, in the following text all data deals with constructed wetlands as vegetated beds unless stated.

Table 5-12. List of countries from which the results were included in the database.

Continent	Country
Europe	Austria, Belgium, Denmark, Croatia, Czech Republic, Estonia, France, Germany, Ireland, Italy, Lithuania, Netherlands, Norway, Poland, Portugal, Slovenia, Spain, Sweden, Switzerland, United Kingdom
North America	Canada, Mexico, USA
Central and South America	Brazil, Colombia, Costa Rica, Ecuador, Jamaica
Asia	China, India, Israel, Japan, Jordan, Korea, Nepal, Taiwan, Thailand
Africa	Egypt, Morocco, South Africa, Tanzania
Australia and Oceania	Australia, New Zealand

Table 5-13. Number of entries for major types of wastewaters.

Type of wastewater	Number of entries								
	COD	BOD_5	TSS	TP	TN	NH_4-N	NO_x-N	TKN	orgN
Agricultural*	39	45	57	45	32	46	33	17	14
Industrial**	40	48	37	10	18	46	29	35	14
Landfill leachate	7	25	8	11	8	25	8	5	5
Municipal and domestic	834	1,318	1,275	779	450	1,012	566	270	173
Other***	2	2	2	8	3	9	8		

*pig farm, goat farm, chicken manure, dairy farm, shrimp aquaculture, fish farm
**food processing, meat processing, potato starch production, cheese production, abattoir (slaughterhouse), distillery, winery, tannery, petrochemical, mixed
***stormwater runoff, greenhouse runoff, polluted river

In Table 5-14, removal of various pollutants in common pretreatment units is shown. It is obvious that for organics and suspended solids removal effect of these units may be substantial. Therefore, it is important to maintain pretreatment units properly. Removal of total nitrogen and phosphorus is very low in pretreatment units. Removal of total nitrogen in pretreatment units occurs mostly due to reduction of nitrate because pretreatment units are usually anoxic/anaerobic. However, municipal sewage contains negligible concentrations of nitrate. Ammonia concentrations after the passage through pretreatment units does not change too much, in some cases the concentration even increases due to conversion of organic nitrogen to ammonia. Concentrations of organic nitrogen usually decrease because ammonification process proceeds under both aerobic and anaerobic conditions (see section 2.3.1). Retention of phosphorus in pretreatment units occurs mostly due to sedimentation of particulate fraction.

Table 5-14. Removal of pollutants in common pretreatment units (septic and Imhoff tanks) used in constructed wetland technology. Data from systems treating municipal and dairy wastewaters in the Czech Republic, Lithuania, Poland and Mexico. n=number of annual means.

Parameter	Raw wastewater (mg l^{-1})	After pretreatment (mg l^{-1})	Average efficiency (%)	n
BOD_5	211	101	40.1	96
COD	481	233	37	95
TSS	231	72.5	45.6	88
TP	10.1	7.8	10.5	83
TN	60	50.9	8.8	59
NH_4-N	34.8	34.8	-1.8	61
NO_x-N	3.26	2.48	11.9	57
N_{org}	17.1	10.3	25.8	43
TKN	47.6	45.5	-0.8	38

5.4.1 Organics

Organic matter is decomposed in HF constructed wetlands by both aerobic and anaerobic microbial processes (see Section 2.2) as well as by sedimentation and filtration of particulate organic matter. Because of heavy loading and continuous saturation of the filtration bed anoxic/anaerobic processes prevail while aerobic processes are restricted to small zones adjacent to roots and rhizomes (radial oxygen loss) and to thin surface layer where oxygen diffusion from the atmosphere may occur. In lightly-loaded systems dissolved oxygen may be also carried out by inflowing wastewater.

In Tables 5-15 and 5-16, evaluation of BOD_5 and COD data in the database (see section 5.4 above) is shown. The majority of results have been obtained from systems treating municipal/domestic wastewaters while considerably less information exists on other types of wastewater. However, it was not surprising that the highest inflow concentrations and loadings were recorded for agricultural and industrial wastewaters.

The results presented in Table 5-15 indicate that HF constructed wetlands can treat wastewaters with very wide range inflow BOD_5 concentrations as well as wide range of inflow loadings. For municipal sewage the results are in a good agreement with other review information. For example, Puigagut et al. (2007) reported in their review the average inflow/outflow concentrations in the range of 27.5 to 173 mg l^{-1} and 5.9 to 32 mg l^{-1}, respectively. The data were taken from 201 systems in 8 countries (Czech Republic, Denmark, Germany, Poland, Slovenia, Spain, Sweden, USA) treating mostly municipal sewage. However, we believe that it is useful to distinguish between low loaded (tertiary) and secondary systems.

Table 5-15. Removal of BOD_5 in HF constructed wetlands for various types of wastewater. For BOD_5, the level of 40 mg l^{-1} was selected to distinguish tertiary and secondary levels. Rem = removed load. *n= number of entries (annual means) with number of systems in parentheses.

	Concentration (mg l^{-1})		**Eff. (%)**	**n***	**Loading (kg BOD_5 ha^{-1} d^{-1})**			**n***
	In	**Out**			**In**	**Out**	**Rem**	
All results	170	42	73.4	1,143 (438)	116	37	79	966 (368)
Municipal (<40 mg l^{-1})	19.5	6.8	60.7	281 (122)	49.3	17.2	32.1	230 (103)
Municipal (>40 mg l^{-1})	178	32	80.7	746 (261)	97	19.4	77.6	624 (213)
Agriculture	464	183	68.2	43 (19)	541	294	246	43 (18)
Industry	652	254	60.1	48 (23)	353	158	195	45 (22)
Landfill leachate	155	96	32.8	25 (13)	28	17.1	10.9	24 (12)

Table 5-16. Removal of COD in HF constructed wetlands for various types of wastewater. Rem = removed load. *n= number of entries (annual means) with number of systems in parentheses.

	Concentration (mg l^{-1})		**Eff. (%)**	**n***	**Loading (kg COD ha^{-1} d^{-1})**			**n***
	In	**Out**			**In**	**Out**	**Rem**	
All results	427	143	62.7	641(292)	371	163	208	578(264)
Municipal	287	76	63.2	556(244)	237	88	149	493(217)
Agriculture	871	327	63.0	38(17)	1,239	602	637	37(17)
Industry	1,856	789	63.1	40(25)	1,212	652	560	40(24)
Landfill leachate	933	698	24.9	7(6)	330	279	51	7(6)

First-order degradation represents the basic design equation employed widely (Reed et al., 1988; Reed, 1993; Cooper et al., 1996; Vymazal et al., 1998b; Mitchell et al., 1998; Dahab et al., 2001). This approach has been used for design and to predict removal performance for basically all pollutants of interest in constructed wetlands. Its inadequacies has been recognized (Bavor et al., 1988, Kadlec and Knight, 1996) but it is still seen as the most appropriate design equation describing pollutant removal in light of present knowledge (Kadlec et al., 2000). Mitchell and McNevin (2001) suggested the model based on the assumption that the biological processes in wetlands, like other biological systems, exhibit Monod kinetics. A Monod approach fits well with observed wetland performance. It predicts first-order behavior at low concentrations, that is, pollutant removal rates which increase with increasing pollutant concentration; and zero-order or saturated behavior at high pollutant concentration, that is, a maximum pollutant removal rate.

Recently, more complex models have been presented (Wynn and Liehr, 2001; Mashauri and Kayombo, 2002; Langergraber, 2003; Rousseau et al., 2004a; Marsili-Libelli and Checchi, 2005). However, most of these models require many parameters which are often difficult to measure and also include many presumptions which may not be valid for all systems.

The regression equations based on inflow and outflow concentrations usually do not provide a good fit. Important factors such as flow, climate, bed material, bed design (length, width, depth), are neglected, leading to a variety of regression equations (Rousseau et al., 2004a). Schierup et al. (1990b) reported for 69 Danish HF constructed wetlands the relationship between inflow and outflow BOD_5 concentrations

$$C_{out} = 0.06\ C_{in} + 5.37\ (R^2 = 0.38) \tag{5.13}$$

where C_{in} and C_{out} are inflow and outflow BOD_5 concentrations (mg l^{-1}).

Brix (1998) reported similar relationship for 90 Danish HF systems

$$C_{out} = 0.03\ C_{in} + 6.1\ (R^2 = 0.08) \tag{5.14}$$

Vymazal (2001c) reported the relationship for BOD_5 in 23 HF systems in the Czech Republic:

$$C_{out} = 0.09\ C_{in} + 2.71\ (R^2 = 0.33) \tag{5.15}$$

The poor fit between inflow and outflow BOD_5 concentrations has also been reported from other countries. Data presented by Urbanc-Berčič et al. (1998) yielded slightly better correlation ($R^2 = 0.41$) but results presented by Kowalik and Obarska-Pempkowiak (1998) and Kalisz and Salbut (1995) yielded a very loose relationship ($R^2 = 0.044$) for Polish HF systems. An even lower value of $R^2 = 0.03$ can be calculated for HF CWs included in the North American database (Kadlec and Knight, 1996).

Inflow-outflow relationships obtained from our database (Fig. 5-72) indicates better fit for secondary systems, i.e., systems with inflow BOD_5 concentration > 40 mg l^{-1}. Better fit between inflow and outflow concentrations was obtained for COD (Fig. 5-72). The better fit for COD as compared to BOD_5 has already been reported, for example by Urbanc-Berčič

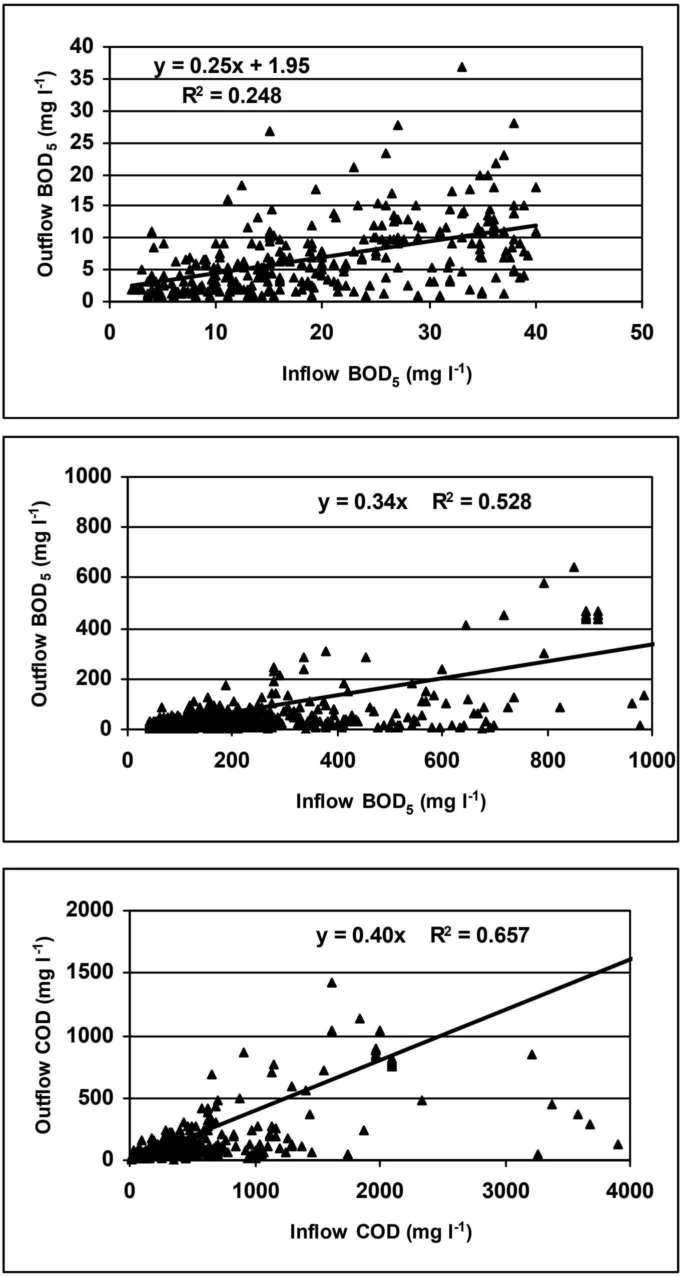


Figure 5-72. Relationship between inflow and outflow BOD_5 and COD concentrations (annual means) in HF constructed wetlands. Top: inflow $BOD_5 \leq 40$ mg l^{-1} (n=300); middle: inflow $BOD_5 > 40$ mg l^{-1} (n=843, maximum inflow concentration: 2,958 mg l^{-1}, highest 19 inflow concentrations not shown in the graph); bottom: COD (n= 641, inflow concentration range of 4.3 to 8,420 mg l^{-1}, highest four values not shown in the graph).

et al. (1998) in Slovenia (R^2 = 0.47), Börner et al. (1998) for HF systems in Germany-Bavaria (R^2 = 0.52) or Vymazal (2001c) in the Czech Republic (R^2 = 0.69). However, detailed analysis of the data revealed that for inflow COD concentrations up to 200 mg l^{-1} the regression is much weaker (R^2= 0.31, n=261). Also the results revealed that with increasing inflow concentrations the relationships with outflow concentrations are getting stronger. For example, for the inflow range up to 500 mg l^{-1} the R^2= 0.41, (n=518) and for inflow between 2,000 and 8,420 mg l^{-1} the R^2 is 0.62 (n=18).

The relationships between inflow and outflow BOD_5 and COD loadings (Fig. 5-73) provide much stronger fit than that for concentrations. Vymazal (2001c) reported for the Czech HF constructed wetlands the following relationship:

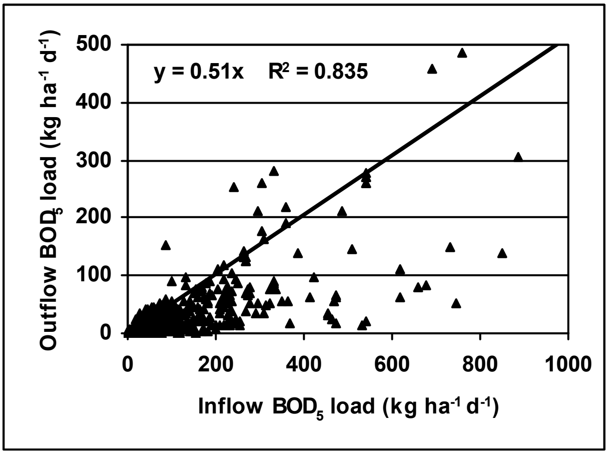
$$L_{out} = 0.13\ L_{in} + 0.56\ (R^2 = 0.53) \tag{5.16}$$

where L_{in} and L_{out} are inflow and outflow BOD_5 loadings (kg ha^{-1} d^{-1}). Using the data from Danish (Schierup et al., 1990a) and North American (Reed 1993; Kadlec and Knight, 1996) databases, similar relationships also show good correlation (R^2 = 0.71 and 0.67, respectively). However, the most frequent BOD_5 and COD inflow loading range (up to 200 and 500 kg ha^{-1} d^{-1}, respectively) provide less strong relationship (R^2=0.54, n= 848 and R^2=0.50, n=494 for BOD_5 and COD, respectively).

Table 5-17 summarizes the evaluation of k_A vales for various types of wastewater. For BOD_5, the highest value was recorded for tertiary treatment systems, followed by agricultural systems. The lowest value was recorded for landfill leachate systems. Similar situation was found for COD values. For discussion see Section 5.2.6.

Table 5-17. Values of k_A obtained for various types of wastewater. *Only one value for all concentrations.

	BOD_5			**COD**		
	(m d^{-1})	**(m yr^{-1})**	**n**	**(m d^{-1})**	**(m yr^{-1})**	**n**
Municipal (<40 mg l^{-1})	0.298	109	230			
Municipal (>40 mg l^{-1})	0.122	45	624	0.104*	38	494
Agriculture	0.182	66	43	0.655	239	37
Industry	0.061	22	45	0.155	57	40
Landfill leachate	0.012	4.4	24	0.100	37	7
Total	0.160	58	967	0.136	50	578



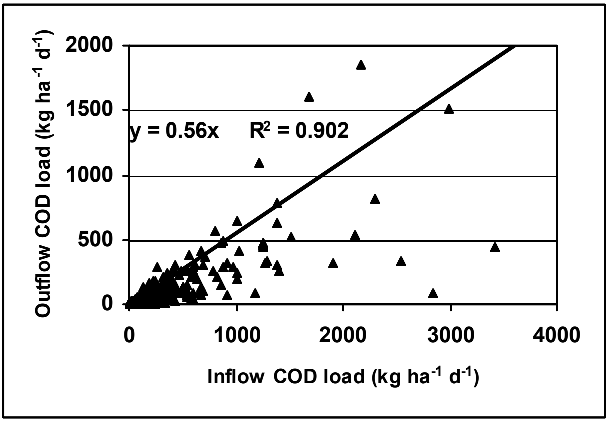


Figure 5-73. Relationship between inflow and outflow BOD_5 (top, n=988, inflow range. 0.3 – 8,580 kg ha^{-1} d^{-1}) and COD (bottom, n=579, inflow range 3.3 – 14,769 kg ha^{-1} d^{-1}) loadings in HF constructed wetlands. Highest 12 values in both charts are not shown.

The highest removal of organics takes place within several meters of the inflow zone and further decrease in concentrations is much smaller (e.g., Bavor et al., 1987; Vymazal, 2003b; Fonder and Xanthoulis, 2007, see also Fig. 5-74). It is important to realize that substantial removal of organics may take place even in the distribution zone filled with large stones (Fig. 5-74) and therefore, it is always necessary to include the distribution zone area into the total surface area of the vegetated beds.

Aguirre et al. (2005) found that the shallow HF wetlands (0.27 m water depth) removed more COD (72-81%), BOD_5 (72-85%) than parallel deep HF wetlands (0.6 m water depth) where COD and BOD_5 removals varied between 59-64%, 51-57%, respectively. The higher removal of organics was explained by more aerobic conditions in the shallow beds. However, the authors pointed out that this finding cannot be generalized because the

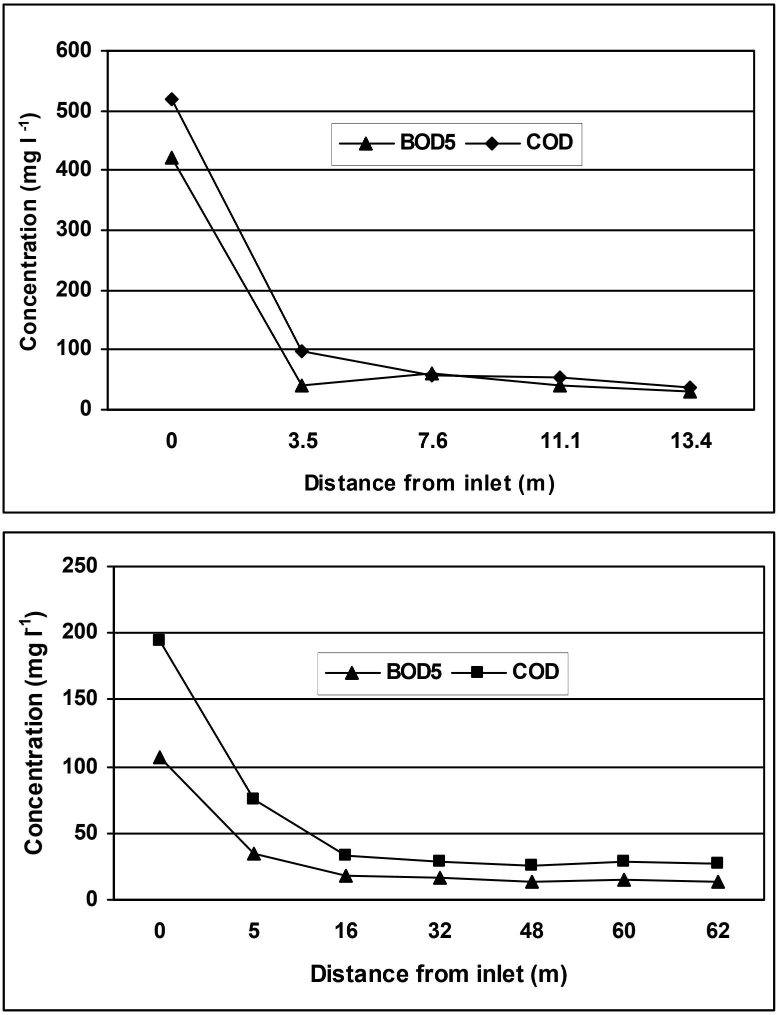


Figure 5-74. Removal of organics along the longitudinal profile of the vegetated beds of HF constructed wetlands. Data from Belgium (top, Fonder and Xanthoulis, 2007) and the Czech Republic (bottom, Vymazal, 2003b). The last distances represent the final outflow. In the bottom chart, the distance between 0 and 5 meters represents the distribution zone.

efficiency can depend, among other factors, on aerial organic loading rate used and the quality of the organic matter. On the other hand, Bayley et al. (2003) reported no significant difference in BOD attenuation rate between the upper, middle and bottom layers within the 0.5 m deep pilot-scale constructed wetland planted with *Phragmites australis* and filled with 10 mm diameter gravel in Lismore, NSW, Australia. Vymazal and Maša (2003) reported on the use of pulsing water level in HF constructed wetland at Dolní Město, Czech Republic. The pulsing (8 and 15 cm) was intended to increase ammonia removal through oxygenation of the bed but the results indicated more positive effect on removal BOD_5 and COD than ammonia.

The influence of vegetation removal of organic matter in HF constructed wetlands is discussed in Section 5.4.7.

5.4.2 Suspended solids

Suspended solids in waters are defined by the method of analysis. Generally, suspended solids are defined as those solids retained on a standard glass fiber filter that typically has a nominal pore size of 1.2 μm. The type of filter holder, the pore size, porosity, area and thickness of the filter, and the amount of material deposited on the filter are the principal factors affecting the separation of suspended from dissolved solids. As a result, the measurement reported for total suspended solids may include particle sizes ranging from greater than 100 μm to about 1 μm. Soluble (dissolved) solids would therefore include colloidal solids smaller than 1 μm and molecules in true solution. A classical method for solids classification by size would include the following (U.S. EPA, 2000):

- settleable solids (> 100 μm)
- supracolloidal solids (1-100 μm)
- colloidal solids (10^{-3}-1 μm)
- soluble solids (< 10^{-3} μm)

As pretreatment units used for HF constructed wetlands are generally able to remove only particles > 40 μm (Table 5-1) wastewater entering the filtration beds of constructed wetlands always contains suspended solids which are trapped within the filtration material. However, the composition of suspended solids is quite different based on the type of wastewater (U.S. EPA, 2000).

One of the primary removal/retention mechanisms for suspended solids in HF constructed wetlands is the flocculation and settling of colloidal and supracolloidal particulates. Other effective removal mechanisms in HF systems are gravity sedimentation (discrete and flocculant), straining and physical capture and adsorption on biomass film attached to gravel and root systems (see section 5.4.5 for details on straining and adsorption).

Sapkota and Bavor (1992) summarized that retention of particles by straining is possible when the ratio of diameter of particles to be removed to the diameter of the filtration medium is greater than or equal to 0.05. Also, the authors pointed out that gravity settling plays a significant role only in the capture of larger particles having sizes greater than 5 μm; then the removal efficiency is proportional to the square of the diameter.

The entrapment of suspended solids in HF constructed wetlands is the highest within the area where distribution zone filled with large stones meets with the filtration bed material which has much smaller grain size. It has also been shown that most suspended solids are retained within the first several meters of the bed (e.g. Bavor et al., 1987, 1989; Davies and Cottingham, 1992; Vymazal, 2003b). Nguyen (2001) reported that clogging of the pore spaces over a 5-year wetland operation was due to accumulation of refractory organic solids, particularly in the top 100 mm of the gravel bed. Microbial respiration rate and microbial biomass were significantly correlated with stable organic matter fractions, suggesting that these microbial parameters may be used to predict the nature of accumulated organic matter fractions. Ouellet-Plamondon et al. (2006) reported that artificial aeration slightly enhanced TSS removal in all seasons regardless of treatment, probably by maintaining empty space in the head part of the gravel bed.

Mwegoha et al. (2002) developed a model to predict the effect of suspended solids accumulation on hydraulic conductivity of gravel based HF constructed wetland. The results of the model showed that the reduction of hydraulic conductivity of the medium was highly attributed to the capture of non-degradable solids, accounting for 98.4% of the total effect. Kimwaga et al. (2002b) reported that filtration process was the major route of TSS removal in a HF constructed wetland accounting for 75% removal followed by biodegradation, which accounted for 15%.

In Table 5-18, results of the survey are presented. The vast majority of available results were obtained from systems treating municipal wastewater. The highest inflow concentrations and loadings were recorded in agricultural systems with loadings being one order of magnitude higher as compared to other types of wastewater. The average k_A values for various types of wastewaters were: all results: 0.25 m d^{-1}, municipal: 0.19 m d^{-1}, agricultural: 1.30 m d^{-1}, industrial: 0.16 m d^{-1} and landfill leachate: 0.022 m d^{-1}. All the values are much lower as compared to value of 2.74 m d^{-1} proposed by Kadlec and Knight (1996).

Table 5-18. Removal of TSS in HF constructed wetlands for various types of wastewater. Rem = removed load. *n= number of entries (annual means) with number of systems in parentheses.

	Concentration (mg l^{-1})		Eff. (%)	n*	Loading (kg TSS ha^{-1} d^{-1})			n*
	In	Out			In	Out	Rem	
All results	141	35.1	68.6	1,076(367)	191	77	114	927(314)
Municipal	113	22.3	68.1	975(319)	111	28	83	828(271)
Agriculture	516	180	76.9	56(26)	1,430	779	637	54(23)
Industry	239	128	71.6	37(17)	212	137	75	37(15)
Landfill leachate	391	86	54.5	8(5)	40	17	23	8(5)

Similarly to organics, the most intensive retention of suspended solids takes place in the inflow zone (Fig. 5-75). As a consequence this part of the bed usually gets clogged first. For details on clogging see Section 5.2.3.3. The results shown in Figure 5-75 also reveal that retention of suspended solids takes place in the distribution zone filled with large stones even before the water reaches the filtration bed.

The relationship between inflow and outflow TSS concentrations is shown in Figure 5-76. The relationship for inflow TSS concentrations < 200 mg l^{-1} yielded less strong regression (R^2 = 0.285). However, this regression is much stronger than those reported by Brix (1998) for 90 HF constructed wetlands in Denmark (R^2 = 0.03, see also Fig. 7-11). On the other hand, Brix (1994b) reported a comparable regression (R^2= 0.45) for 77 HF systems in Denmark and United Kingdom. In Figure 5-77, relationship between inflow and outflow TSS loadings is shown. The range of inflow values is very wide, however, the regression is quite strong also for the most frequently occurring inflow loading range up to 500 kg TSS ha^{-1} d^{-1} (n=880, R^2=0.60). Brix (1994b) found R^2=45 for 51 systems in Denmark and United Kingdom.

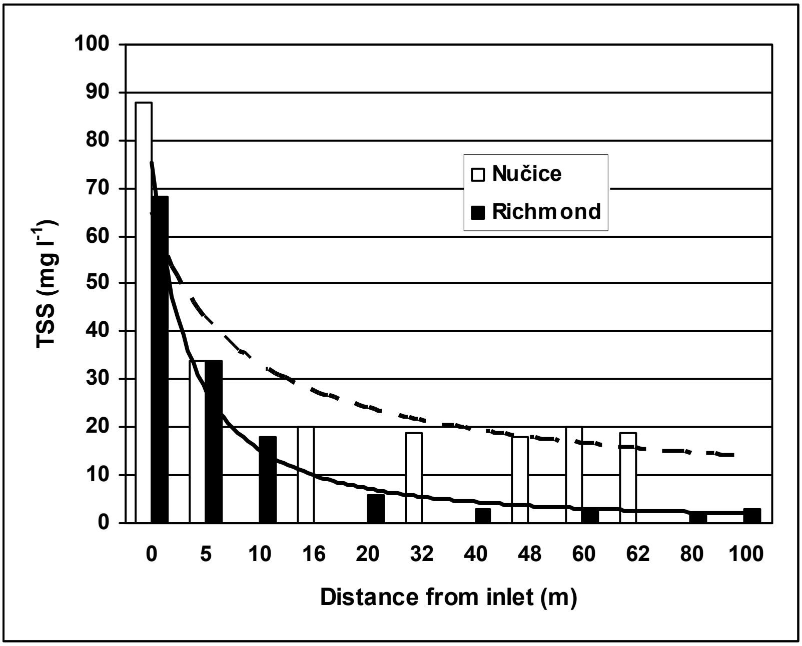


Figure 5-75. Removal of TSS along the longitudinal profile of HF constructed wetlands Nučice, Czech Republic (data from Vymazal, 2003b, dashed line) and Richmond, NSW, Australia (data from Bavor et al., 1987, solid line). In Nučice, the distance between 0 and 5 meters represents the distribution zone.

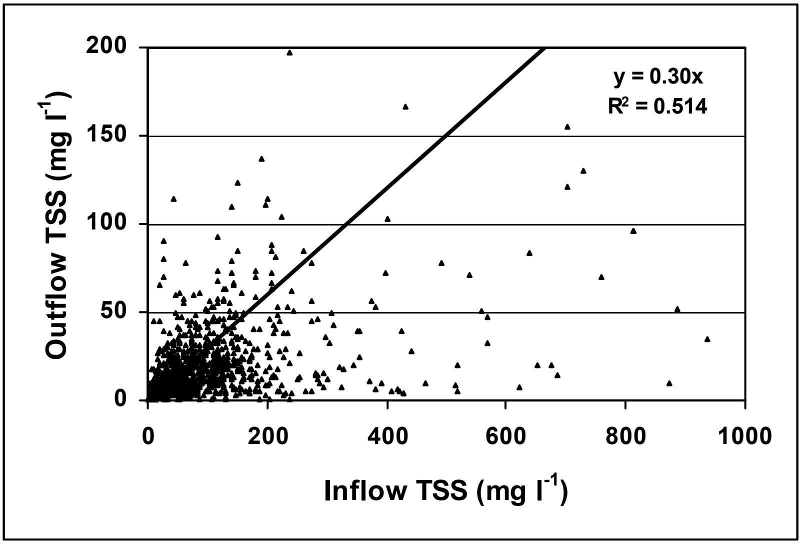


Figure 5-76. Relationship between inflow and outflow TSS concentrations in HF constructed wetlands (n=1,022, maximum inflow value 4,200 mg l^{-1}, highest 17 values not shown).

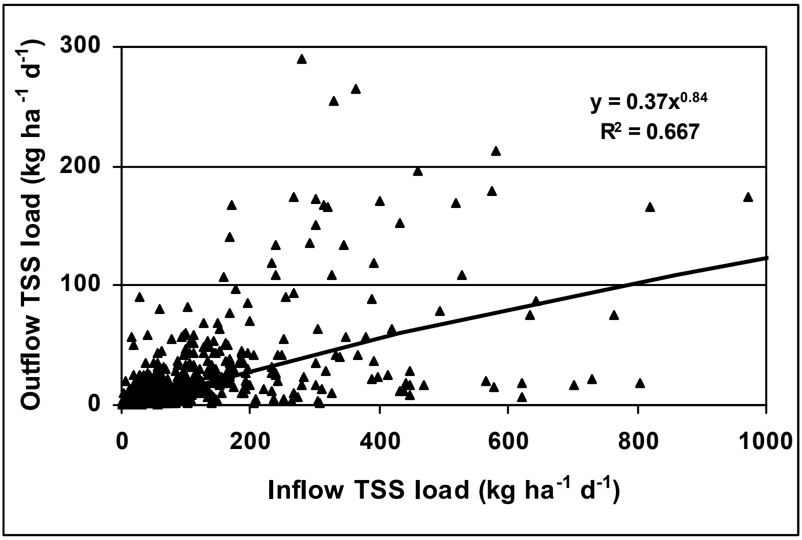


Figure 5-77. Relationship between inflow and outflow TSS loadings in HF constructed wetlands (n=928, inflow range 0.1 – 33,500 kg ha^{-1} d^{-1}). Highest 18 values over 1000 kg ha^{-1} d^{-1} not shown.

5.4.3 Nitrogen

Transformations of nitrogen in wetland systems are manifold (Section 3.2) but removal pathways which literally remove nitrogen from the system are only few: denitrification, volatilization, and plant uptake if connected with harvesting. Other mechanisms, such as nitrification or ammonification, are only responsible for transformations among various nitrogen forms or others retain nitrogen in the beds (adsorption, burial). Vymazal (2007) summarized that in HF constructed wetlands the nitrogen-removing mechanisms are quite limited due to lack of oxygen in filtration beds due to continuous waterlogging of the bed (see Section 2.1) and absence of free water column. The lack of oxygen in vegetated beds restricts nitrification, i.e. oxidation of ammonia which is the major form of nitrogen in most wastewaters.

Volatilization is very limited by the fact that HF CWs do not have free water surface and algal activity, which would increase the pH value of water, is negligible in these systems. The fine-grained soils always show better nitrogen removal through adsorption than the coarse-grained soil (Geller et al., 1990). The higher elimination rate can be explained by the higher cation exchange capacity of the fine-grained soils. However, fine-grained soils are not used for HF systems, at present, because of poor hydraulic conductivity. Therefore, the adsorption capacity of the commonly used media (pea gravel, crushed rock) is usually very limited. However, Zhu and Sikora (1994) found that adsorption of NH_4 onto the gravel was high and resulted in 40% removal in microcosm experimental wetlands.

Brix and Schierup (1989c) suggested that standing stock in aboveground biomass of emergent macrophytes, and which is thus available for harvesting, is roughly between 20 and 250 g N m^{-2}. Vymazal (1995a) reported aboveground N standing stock in the range of 22 to 88 g N m^{-2} for 29 various emergent species. Johnston (1991) gives the range for nitrogen standing stock in emergent species between 0.6 and 72 g N m^{-2} with an arithmetic mean of 20.7 g N m^{-2}. Mitsch and Gosselink (2000) reported that the aboveground stock of nitrogen in freshwater marsh plants ranges from as low as 3 to 29 g N m^{-2}. Vymazal et al. (1999) reported nitrogen standing stock in aboveground biomass of *Phragmites australis* and *Phalaris arundinacea* growing in natural stands in the range of 0.04 to 63.4 g N m^{-2} and 2.0 to 15.5 g N m^{-2}, respectively.

Kuusemets et al. (2002) and Mander et al. (2004) reported nitrogen accumulation in plant biomass (*Phragmites australis* and *Scirpus sylvaticus*) growing in a HF constructed wetland at Kodijärve, Estonia to be 13.7 and 67.6 g N m^{-2} in 2001 and 2002, respectively. The difference between years was influenced by much higher biomass in 2002 as compared to 2001 (2,143 and 621 g DM m^{-2}, respectively). The biomass included above- and belowground parts and litter. Most of the nitrogen standing stock was allocated belowground. Tanner (2001) found mean combined above (including standing dead) and below-ground *Scirpus tabernaemontani* nitrogen accumulation between 48 and 69 g N m^{-2} increasing with wetland wastewater loading.

However, the amount of nitrogen in harvestable biomass must be evaluated using the standing stock in biomass after the harvest. The results obtained in a short term experiments could be very misleading and incorrect. Okurut (2001) observed the nitrogen uptake rate during the exponential growth of *Phragmites mauritianus* in HF system in Uganda to be 1.04 g N m^{-2} d^{-1} but during the steady state the uptake rate dropped to only 0.018 g N m^{-2} d^{-1}. Langergraber (2005) in his review found daily uptake rates among wetland plants used in constructed wetlands between 6.0 and 744 mg N m^{-2} d^{-1}. Also Headley et al. (2005) noted in their study in Australia that nitrogen uptake varies greatly during the year. These results reveal that uptake varies greatly with season and growing stage of plants and the results could hardly be used for estimation of nitrogen amount that can be removed via harvesting.

It has been found that multiple harvest may increase the total standing stock due to high nutrient concentrations in young shoots. For example, Kröpfelová and Vymazal (2006) found that the amount of nitrogen removal via multiple harvest of *Phalaris arundinacea* increased by 45% as compared to a single harvest. However, question remains to be answered is whether the multiple harvesting if effective because the amount of nitrogen and other nutrients is very low even with multiple harvest. Unfortunately, there are very few data on multiple harvesting of plants in HF constructed wetlands and there are no data available on costs connected with harvesting and subsequent biomass handling.

Also, Vymazal (2005a) in his review in nitrogen removal via plant harvesting pointed out that harvest of aboveground biomass of emergent macrophytes in treatment wetlands does remove some nutrients. However, the annual amounts of removable nutrients are low (Table 5-19) as compared to common inflow loadings to constructed wetlands (Table 5-20). Nitrogen removal is limited to a maximum about 50-60 g N m^{-2} at the peak standing stock which constitutes commonly less than 10% of the inflow nitrogen load. However, in temperate and cold regions some plants (e.g. *Phragmites australis*) cannot be harvested during the peak standing stock. Also, in temperate and cold climates macrophyte biomass is needed for surface insulation and therefore, biomass could be harvested only at the end of the winter when nitrogen content in the biomass is substantially lower as compared to summer.

The amount of nutrients stored in the aboveground plant biomass varies only little with the inflow loading in wetlands treating wastewaters and, therefore, the relative extent and importance of nutrient removal via plant harvesting depends on the loading of the wetland. Higher nitrogen removal via harvesting could be expected only for lightly-loaded wetlands (Vymazal, 2005a). Tanner (2001), for example, reported that nitrogen standing stock in *Schoenoplectus validus* increased from 23 to only 40 g N m^{-2} when inflow load increased from 421 to 1,256 g N m^{-2} yr^{-1}. The respective removal percentage dropped from 5.3% to only 3.2%. For further data on nitrogen removal via harvesting see also Kadlec (2005).

Table 5-19. Examples of nitrogen standing stock (SS, g N m^{-2}) in aboveground biomass of plants growing in HF constructed wetlands treating municipal sewage (unless mentioned) and comparison with inflow total nitrogen loading (g N m^{-2} yr^{-1}) and SS proportion of the total removed nitrogen in the system.

Plant species	**Location**	**SS**	**SS as % of inflow load**	**% of removed nitrogen**	**Reference**
Phragmites australis	Poland	58.7	1.2	8.3	1
Phragmites australis	Czech Rep.	47.2	4.1	9.3	2
Phragmites australis	Poland	46.8	6.9	12.5	3
Phragmites australis	Australia	43.8	21	27.4	4
Schoenoplectus validus	New Zealand	40	3.2		5
Baumea articulata	Australia	34	3.9		6
Phragmites australis	Poland	32.6	0.5	5.5	1
Carex fascicularis	Australia	30	3.4		6
Phragmites australis	Poland	29.2	4.7	10.2	3
Phragmites mauritianus	Tanzania	17	3.2	6.7	7
Phalaris arundinacea	USA	16.2	46.8		8
Phragmites australis *Scirpus sylvaticus*	Estonia*	13.7	8.0		9
Phalaris arundinacea	Czech Rep.	12.2	0.9	1.3	2
Phragmites australis	Austria	10.7	2.8	4.1	10
Phylidrum lanuginosum	Australia	5.3	0.6		6

1-Obarska-Pempkowiak (1999), 2-Vymazal (1999c), 3-Obarska-Pempkowiak and Gajewska (2003), 4-Headley (2004), nursery runoff, 5-Tanner (2001), 6- Browning and Greenway (2003), 7-Senzia et al. (2002), 8-Bernard and Lauve (1995), landfill leachate, 9- Kuusemets et al. (2002) *includes aboveground, belowground and litter, 10-Haberl and Perfler (1990)

Senzia et al. (2002) modeled nitrogen transformations and removal in HF constructed wetland planted with *Phragmites mauritianus* in Tanzania. The model was presented as first-order kinetics, except plant uptake and nitrification. Denitrification accounted for 15% removal of the inflow nitrogen load, harvesting of plants removed 13.4%, accretion of organic matter was the major pathway accounting for 19.2% of all the influent nitrogen. Mander et al. (2003b) reported that the most important flux in N budget in HF constructed wetland at Kodijärve, Estonia was N_2 emission via denitrification (41.7%), followed by microbial immobilization (29.2%), accumulation in soil (14.1°%), assimilation by plants (13.8%) and N_2O emission (1.2%).

Tanner and Kadlec (2003) observed in a cascade mesocosm study average rates of N mineralization ranging from 0.22 to 0.53 g m^{-2} d^{-1}, nitrification 0.56 to 2.15 g m^{-2} d^{-1}, denitrification 0.47 to 1.99 g m^{-2} d^{-1} and plant assimilation 0.28 to 0.47 g m^{-2} d^{-1}. Obarska-Pempkowiak and Gajewska (2003) observed in HF wetlands in Wiklino, Poland ammonification, nitrification and denitrification rates between 0.13 and 0.15 g m^{-2} d^{-1}, 0.35 and 0.68 g m^{-2} d^{-1} and 0.51 and 0.67 g m^{-2} d^{-1}, respectively.

Removal of total nitrogen (TN) and various forms of nitrogen in HF constructed wetlands is presented in Tables 5-20 to 5-24. It is obvious that major form of nitrogen in all types of wastewaters is ammonium-N (Tables 5-20 and 5-21). However, organic nitrogen may be an important constituent of total nitrogen, especially in agricultural and industrial wastewaters (Table 5-24). Inflow concentrations of oxidized nitrogen (nitrate- and nitrite-N) are generally low, only industial wastewaters and landfill leachate exhibit elevated concentrations (Table 5-22).

Table 5-20. Removal of TN in HF constructed wetlands for various types of wastewater. Rem = removed load. *n= number of entries (annual means) with number of systems in parentheses.

	Concentration ($mg\ l^{-1}$)		Eff. (%)	n*	Loading ($g\ N\ m^{-2}\ yr^{-1}$)			n*
	In	Out			In	Out	Rem	
All results	63.1	36.0	39.7	476(208)	1,158	738	420	445(188)
Municipal	53.0	29.8	39.4	419(182)	945	581	364	388(162)
Agriculture	116	57.5	51.3	31(13)	2,480	1,531	949	31(13)
Industry	138	102	27.8	18(8)	3,080	2,333	747	18(8)
Landfill leachate	211	126	33.1	8(5)	1,579	1,381	198	8(5)

Table 5-21. Removal of NH_4^+-N in HF constructed wetlands for various types of wastewater. Rem = removed load. *n= number of entries (annual means) with number of systems in parentheses.

	Concentration ($mg\ l^{-1}$)		Eff. (%)	N*	Loading ($g\ NH_4\text{-}N\ m^{-2}\ yr^{-1}$)			N*
	In	Out			In	Out	Rem	
All results	36.1	22.1	22.6	905(305)	824	520	304	825(274)
Municipal	28.4	17.1	21.1	789(254)	663	469	194	711(225)
Agriculture	71.5	39.6	33.8	45(18)	2,722	695	2,027	45(18)
Industry	65.2	48.6	28.0	46(22)	1.246	985	261	43(21)
Landfill leachate	162	98	38.7	25(11)	1,177	791	386	26(10)

Table 5-22. Removal of oxidized nitrogen (NO_x-N) in HF constructed wetlands for various types of wastewater. Rem = removed load. *n= number of entries (annual means) with number of systems in parentheses.

	Concentration ($mg\ l^{-1}$)		Eff. (%)	n*	Loading ($g\ NO_x\text{-}N\ m^{-2}\ yr^{-1}$)			n*
	In	Out			In	Out	Rem	
All results	8.0	5.9	-407	563(210)	643	511	132	528(198)
Municipal	6.5	4.5	-319	494(181)	572	410	162	459(169)
Agriculture	7.5	6.4	-1,871	32(13)	1,931	2,132	-201	32(13)
Industry	16.7	15.1	-437	29(13)	738	398	39	29(13)
Landfill leachate	15.8	11.4	19	8(3)	53	41	12	8(3)

Table 5-23. Removal of TKN in HF constructed wetlands for various types of wastewater. Rem = removed load. *n= number of entries (annual means) with number of systems in parentheses.

	Concentration ($mg\ l^{-1}$)		Eff. (%)	N*	Loading ($g\ TKN\ m^{-2}\ yr^{-1}$)			N*
	In	Out			In	Out	Rem	
All results	64.0	35.9	41.3	316(124)	1,072	652	420	297(120)
Municipal	49.8	26.2	41.5	259(104)	781	437	341	240(100)
Agriculture	161	74.8	56.4	17(8)	2,291	1,177	1,114	17(8)
Industry	124	91.0	31.9	35(10)	2,549	1,930	619	35(10)
Landfill leachate	48.8	21.6	44.8	5(2)	570	334	236	5(2)

Table 5-24. Removal of organic N in HF constructed wetlands for various types of wastewater. Rem = removed load. *n= number of entries (annual means) with number of systems in parentheses.

	Concentration ($mg\ l^{-1}$)		Eff. (%)	n*	Loading ($g\ Norg.\ m^{-2}\ yr^{-1}$)			n*
	In	Out			In	Out	Rem	
All results	20.0	9.0	53.7	192(93)	406	226	180	174(89)
Municipal	12.5	4.9	55.7	159(80)	275	127	148	144(76)
Agriculture	49.6	13.8	62.9	14(6)	658	230	428	14(6)
Industry	76.0	50.1	23.5	14(5)	1,887	1,558	329	11(5)
Landfill leachate	18.8	9.9	49.8	5(2)	210	114	96	5(2)

The relationship between inflow and outflow TN concentrations usually provides good regression (Eqs. 5.17, 5.18 and 5.19). The analysis of our database also yields quite strong regression (Fig. 5-78). The more detailed analysis for the range up to inflow concentration of 100 mg l^{-1} still holds the strong regression (n=402, R^2=0.69). Also, the relationship between inflow and outflow loadings is very strong (Fig. 5-79).

$$C_{out} = 0.42\ C_{in} + 7.68\ (R^2 = 0.72) \tag{5.17}$$

(Vymazal, 1999c; 11 systems in the Czech Republic)

$$C_{out} = 0.49\ C_{in} + 2.5\ (R^2 = 0.58) \tag{5.18}$$

(Brix, 1998; 90 systems in Denmark)

$$C_{out} = 0.67\ C_{in} - 1.23\ (R^2 = 0.85) \tag{5.19}$$

(Brix, 1994b; 73 systems in Denmark and United Kingdom)

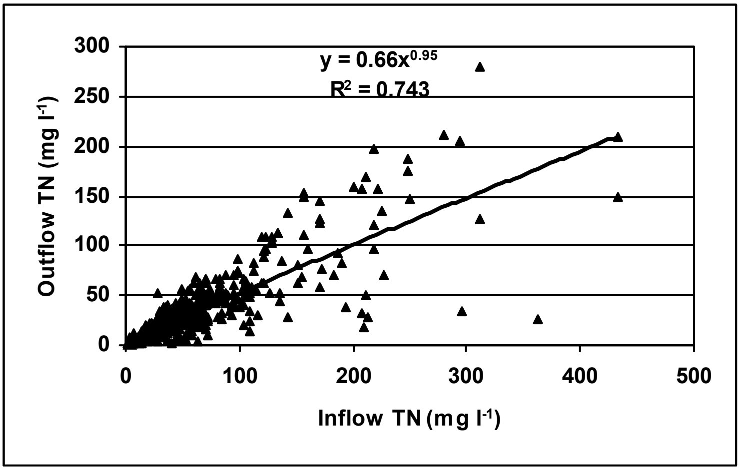


Figure 5-78. Relationship between inflow and outflow TN concentrations in HF constructed wetlands (n=479, inflow range 2 - 433 mg l^{-1}).

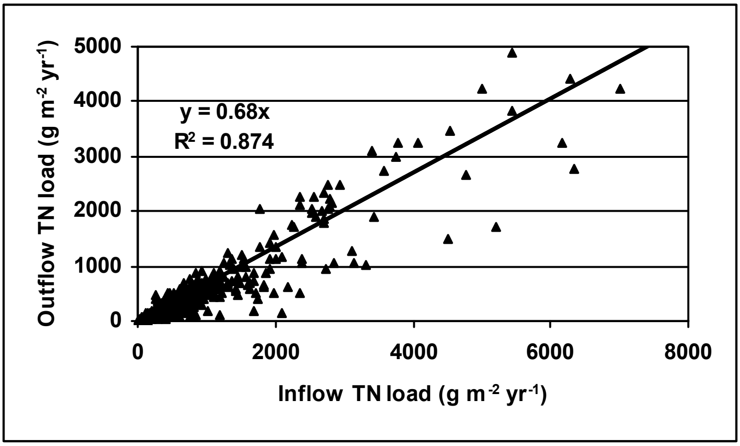


Figure 5-79. Relationship between inflow and outflow TN loadings for HF constructed wetlands (n=447, inflow range 2-16,472mg l^{-1}). Highest 9 inflow values not shown.

Removal of ammonia-N in HF constructed wetlands is low due to very limited nitrification. Results in Table 5-21 indicate that average removal is only about 23% with highest removal in landfill leachate (39%) where inflow concentrations are much higher than in other types of wastewater. In Figures 5-80 and 5-81, the relationship between inflow and outflow ammonia-N concentrations and loads, respectively, are shown.

Platzer and Netter (1992) reported that the influence of temperature on nitrification in HF constructed wetlands is lower than often suggested. The authors suggested that macrophytes and the litter provide an efficient insulation of the bed, thus, reducing the influence of the air temperature. Platzer (1998) suggested that good nitrification in HF constructed wetlands is possible but the bed area necessary is usually extremely large as the maximum load in order to achieve nitrification should not exceed 73 g TKN m^{-2} yr^{-1}. However, this loading rate is very low (see Table 5-23) and for systems which are usually designed for BOD and SS removal (resulting in approximately 5 m^2 per 1 PE) nitrification is hardly achievable.

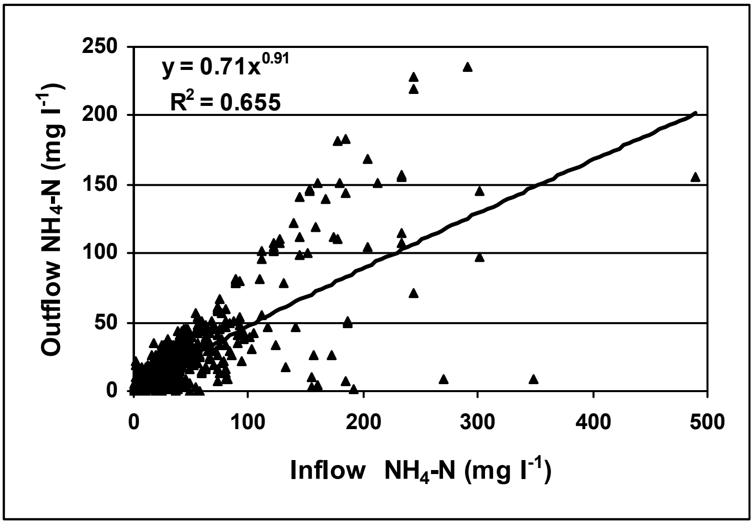


Figure 5-80. Relationship between inflow and outflow NH_4-N concentrations (top, n=913, inflow concentration range 0.1 – 489 mg l^{-1}) in HF constructed wetlands.

Removal of oxidized forms of nitrogen (nitrate-N and nitrite-N) is shown in Table 5-22. The inflow concentrations are mostly low and even the highest average inflow NO_x-N concentrations recorded for industrial wastewaters and landfill leachate are relatively low. The results shown in Table 5-22 indicate that HF constructed wetlands do remove oxidized forms of nitrogen but not completely. The incomplete removal of oxidized nitrogen is often caused by low concentrations of organics necessary for denitrification. The negative values of treatment efficiency in Table 5-22 are caused by the fact many systems receive zero concentrations of oxidized nitrogen and any increase in the outflow may cause large negative efficiencies.

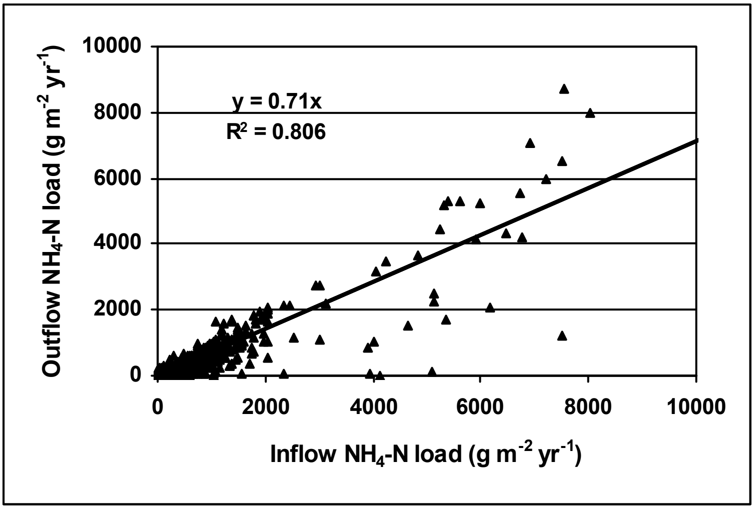


Figure 5-81. Relationship between inflow and outflow NH_4-N loads (bottom, n=827, inflow load range 2-17,634 g m^{-2} yr^{-1}, highest 2 values not shown) in HF constructed wetlands.

In Table 5-25 values of k_A obtained for total nitrogen and various forms of nitrogen is presented. The number of entries for each parameter agrees with the number of entries for loading shown in Table 5-20, 5-21, 5-22 and 5-24. For TN, Kadlec and Knight (1996), based mostly on North American results, reported the average value of 0.074 m d^{-1}. Brix (1998) reported for Danish systems the value of 0.0329 m d^{-1} and Kröpfelová and Vymazal (unpubl.) found the average value of 0.025 m d^{-1} for the Czech systems. For NH_4-N, Kadlec and Knight (1996) reported the value of 0.093 m d^{-1} while Kröpfelová and Vymazal (unpubl.) found the average value of 0.024 m d^{-1}. Kadlec and Knight (1996) reported the k_A values for nitrate-N 0.137 m d^{-1}, the average value for the Czech systems was 0.039 m d^{-1} (Kröpfelová and Vymazal, unpubl).

Table 5-25. Values of k_A obtained for TN and various forms of nitrogen in HF constructed wetlands for various types of wastewater.

	TN		NH_4-N		NO_x-N		N-org.	
	m d^{-1}	m yr^{-1}	m d^{-1}	m yr^{-1}	m d^{-1}	m yr^{-1}	m d^{-1}	m yr^{-1}
All results	0.058	21	0.110	41	0.027	9.9	0.055	20
Municipal	0.033	12	0.048	17.5	0.055	20	0.057	21
Agriculture	0.408	149	1.23	449	-0.339	-124	0.054	20
Industry	0.021	7.6	0.034	12.3	-0.486	-177	0.025	9.1
Landfill leachate	0.0063	2.3	0.014	5.1	0.0037	1.3	0.055	20

Bayley et al. (2003) reported that total nitrogen concentration declined steadily along the pilot-scale constructed wetland planted with *Phragmites australis* and filled with 10 mm diameter gravel in Lismore, NSW, Australia. Aguirre et al. (2005) found that the shallow HF wetlands (0.27 m water depth) removed more ammonia (35-56%) than parallel deep HF wetlands (0.6 m water depth) where ammonia removal varied between 18-29%. However, the authors pointed out that this finding cannot be generalized because the efficiency can depend, among other factors, on organic loading rate used and the quality of the organic matter.

Vymazal (1999c) reported that removal of nitrogen in two HF constructed wetlands in the Czech Republic was only slightly affected by season and there was no significant difference between summer and winter removal rates. Ammonification seemed to be the most temperature-dependent process. For more information on seasonal effects see Section 5.4.8.

Ouellet-Plamondon et al. (2006) reported that artificial aeration improved summer and winter TKN removal for unplanted experimental units, but the additional aeration did not fully compensate for the absence of plants, which suggests that the role of macrophytes goes beyond the sole addition of oxygen to the rhizosphere. Artificial aeration also improved TKN removal in planted units, but to a lower extent than for unplanted units. For further information on vegetation effect on nitrogen removal see Section 5.4.7.

5.4.4 Phosphorus

It is important to note that HF constructed wetlands are seldom built with phosphorus being the primary target of the treatment and therefore, materials with relatively low sorption capacity but high hydraulic conductivity such as river gravel or crushed rock are commonly used (Table 5-26). U.S. EPA (2000) even does not recommend the use of HF constructed wetlands when phosphorus is the primary treatment target. However, in some countries such as Norway, discharge limits for phosphorus are very low and therefore, when constructed wetlands are employed the use of special filter materials with high sorption capacity is necessary.

In order to achieve high phosphorus removal it is necessary to select materials with high P adsorption capacity which depends on chemical and physical properties (e.g., Zhu et al., 1997; Drizo, 1998). Such materials may include minerals with reactive Fe or Al hydroxide or oxide groups on their surfaces, or calcareous materials which can promote precipitation of Ca-phosphate (Drizo et al., 1997; Johansson, 1997; Zhu et al., 1997). These mineral material include dolomite, (Pant et al., 2001; Jenssen and Krogstad, 2003; Prochaska and Zouboulis, 2006), limestone (Drizo et al., 1999), wollastonite (Brooks et al., 2000), opoka (Johansson, 1997), bauxite (Drizo et al., 1999), calcite (Brix et al., 2001b; Molle et al., 2002; Arias et al., 2003), apatite (Molle et al., 2005b), marble (Brix et al., 2001b), diatomaceous earth (Brix et al., 2001b), zeolites (Drizo et al., 1999; Anderson and Rosolen, 2000), glauconite-sandstone (Vohla et al., 2005a), iron-rich sands (Stuanes and Nilsson, 1987; Zhu et al., 1997; Mæhlum and Roseth, 2000; Kvarnström et al., 2004), shell sand (Roseth, 2000; Mæhlum and Roseth, 2000; Søvik and Kløve, 2005; Adám et al., 2007), marl (Gray et al., 2000) and shale (Drizo et al., 1997, 1999, 2000). Brix et al. (2001b) pointed out that the most important characteristic of 13 Danish sands determining their P-sorption capacity was their Ca-content.

Aiming to achieve both high hydraulic conductivity and substantial P removal various materials other than mineral have been tested. Among those materials, light weight aggregates (LWA) and industrial by-products are the most promising. LWA is mainly used for construction purposes such as in building blocks and insulation material. The commercial products are, for example, LECA (light weight clay aggregates), Utelite or Filtralite-P (Zhu et al., 1997). Filtralite is produced from clay or shale by running pelletized aggregates through a rotary kiln at 1,200°C. The other LWAs are produced by similar processes. The chemical composition of of the LWAs differs according to their parent material. The Norwegian Filtralite-P uses clay illite with some natural additions (Zhu et al. 1997; Jenssen and Krogstad, 2003). LWAs provide high sorption capacity (Table 5-26) but they are substantially more expensive than natural materials (Zhu et al., 2003).

Industrial by- and waste-products include blast and electric arc furnaces steel slags (Duncan, 1992; Mann and Bavor, 1993; Sakadevan and Bavor, 1998; Anderson and Rosolen, 2000; Grüneberg and Kern, 2001; Silyn-Roberts and Lewis, 2003, Korkusuz et al., 2004, 2005; Xu et al., 2006; Weber et al., 2006) fly ash (inorganic waste product from coal or oil shale combustion, (Mann and Bavor, 1993; Drizo et al., 1999; Vohla et al., 2005a), crushed concrete (Molle et al., 2002, 2003) iron ochre (Heal et al., 2005), and treated wood chips (Eberhardt et al., 2006).

In the literature, there are many references concerning the laboratory experiments aimed at determination of maximum sorption capacity of various filtration materials (Table 5-26). However, it is difficult to compare them as the experiments are carried out under various experimental conditions (e.g., temperature, duration of the test, initial P concentration, P speciation, medium composition, particle size, batch and column set-up) (e.g., Zhu et al., 1997; Sakadevan and Bavor, 1998; Drizo et al., 2002; Westholm, 2006).

Every sorptive material has a finite sorption capacity. However, assessing this capacity is difficult. The use of adsorption isotherms and calculation of an adsorption maximum depends on several assumptions, which are very different from the actual conditions in a constructed wetland (Jenssen and Krogstad, 2003). Several field studies indicated that long-term phosphorus removal in real treatment systems is much lower as compared to laboratory

Table 5-26. P-sorption capacity in mg P kg^{-1} for some natural media, industrial byproducts and manufactured materials potentially usable for HF constructed wetlands.

Material	Sorption capacity (mg P kg^{-1})	References
Natural materials		
Gravels	8-48	Mann (1990), Mann and Bavor (1993), Seo et al. (2005)
Gravels rich in Ca,Mg,Fe,Al	1,200-1,700	Vohla et al. (2005a)
Various “ordinary” sands	14-290	Stuanes and Nilsson (1987), Krogstad and Løvstad (1987), Arias et al. (2001), del Bubba et al. (2003), Xu et al. (2006)
Fe-rich sand	600-1,230	Stuanes and Nilsson (1987), Zhu et al. (1997), Kvarnström et al. (2004), Vohla et al. (2005a)
Shell sand	335-17,000	Roseth (2000), Søvik and Kløve (2005), Adám et al. (2007)
Sandstone	167	Mann (1990)
Zeolite	460-2,200	Drizo et al. (1999), Sakadevan and Bavor (1998)
Bauxite	610	Drizo et al. (1999)
Ground oyster shell	834	Seo et al. (2005)
Serpentine	1,000	Drizo et al. (2006)
Marl	1,184	Gray et al. (2000)
Dolomite	1,443-2,500	Jenssen and Krogsrad (2003)
Apatite	4,760	Molle et al. (2005b)
Calcite	400-8,750	Cole et al. (1953), Molle et al. (2002)
Iron-ore	1,400	Grüneberg and Kern (2001)
Manufactured material		
LWA (Norway)*	209-12,000	Zhu et al. (1997), Drizo et al. (1999), Jenssen and Krogstad (2003), Kvarnström et al. (2004)
LWA (Sweden)	46-565	Zhu et al. (1997)
LWA(USA)	37-3,460	Zhu et al. (1997)
LWA (Estonia)	100-200	Vohla et al. (2005a)
By-products		
Blast furnace granulated slag	50-44,200	Mann and Bavor (1993), Sakadevan and Bavor (1998), Johansson and Gustafsson (2000), Grüneberg and Kern (2001), Anderson and Rosolen, 2000; Korkusuz et al. (2004, 2005), Xu et al. (2006)
Electric arc furnace steel slag	300-3,650	Drizo et al. (2006), Chazarenc et al. (2006), Weber et al. (2006)
Fly ash	260-13,800	Mann and Bavor (1993), Drizo et al. (1999), Vohla et al. (2005a), Cheung and Venkitachalam (2000)
Crushed concrete	6,390	Molle et al. (2002)
Ochre	26,000	Heal et al., 2005

*the first Filtralite generation had usually sorption capacity between 400 and 3,500 mg P kg^{-1}, the recent Filtralite P has capacity up to 12,000 g P kg^{-1} (Ádám et al., 2005).

batch experiments (e.g. Ádám et al., 2007). Jenssen and Krogstad (2003) suggested that a phosphorus removal of 50% of the sorption capacity measured in the laboratory, using 360 mg l^{-1} phosphate solution, should be used for design purposes. Molle et al. (2003) pointed out that batch experiments and Langmuir estimations are not sufficient to define long term P removal in constructed wetlands. They also suggested that crystal growth seemed to be the final P removal mechanism with calcareous materials.

Arias and Brix (2005b) pointed out that equilibrium isotherm is an indicator of the potential binding capacity in full-scale systems. However, the binding capacities are still insufficient for the establishment of external P-removal filter; the volumes of the filters would be too large to be of practical use.

Tanner et al. (1999) studied the accumulation of phosphorus in the substrata and surface deposits of pilot scale gravel-based HF constructed wetlands in New Zealand. Four different hydraulic loading rates of farm dairy wastewaters provided mean cumulative TP loadings ranging from 520 to 2,000 g m^{-2} after 5 years operation. Mean substratum TP accumulation after 2 years ranged from 52 to 100 g m^{-2} in the planted and wetlands and from 40 to 51 g m^{-2} in the unplanted wetlands, both showing a general increase with wastewater loading. After 5 years operation, associated with declining wetland TP removal performance, substrate TP accumulation in all the planted wetlands converged at similar values of 115 to 128 g m^{-2}. This represents 6 to 23% of their cumulative loadings. Vohla et al. (2005b) found that concentrations of phosphorus in the HF constructed wetland at Kodijärve, Estonia increased between 1997 to 2004 from 15 to 118 mg P kg^{-1}. In the cumulative removal of total P in Kodijärve wetland (64 kg in the period 1997-2004), a significant part is formed by the accumulation of lactate-soluble P in the filter material (up to 28.7 kg P). The annual accumulation of P decreased over the monitored period – from 20.7 kg P yr^{-1} in 1997 to 6.6 and 4.5 kg P yr^{-1} in 2002 and 2004, respectively. The annual average phosphorus retention decreased between 1997 and 2002 from 74.5 to 24.7 g P m^{-2} yr^{-1}.

In Waiuku, New Zealand four different media were tested to treat facultative pond effluent: smelter slag (d_{10} = 6.5 mm), greywacke (d_{10} = 3.3 mm), scoria lava (d_{10} = 3.3 mm) and basalt (d_{10} = 6.0 mm). The media did not differ too much in removing BOD, TSS, TKN, and TN. However, the media differed in phosphorus removal. Inflow TP concentration of 7.4 mg l^{-1} was reduced to 2.9 mg l^{-1}, 4.6 mg l^{-1}, 5.9 mg l^{-1} and 6.8 mg l^{-1} for smelter slag, shallow scoria lava, basalt and greywacke, respectively Duncan (1992).

In Table 5-27, removal of phosphorus in HF constructed wetlands from various wastewaters is shown. Average removal efficiency varies between 41% and 66% but outflow concentrations are frequently still quite high. Relationship between inflow and outflow TP concentrations and loads are presented in Figure 5-82.

Table 5-27. Removal of TP in HF constructed wetlands for various types of wastewater. Rem = removed load. *n= number of entries (annual means) with number of systems in parentheses.

	Concentration ($mg\ l^{-1}$)		Eff. (%)	n*	Loading ($g\ TP\ m^{-2}\ yr^{-1}$)			n*
	In	**Out**			**In**	**Out**	**Rem**	
All results	9.6	4.8	42.8	711(272)	263	178	85	572(238)
Municipal	8.7	4.4	40.9	643(247)	247	178	69	509(213)
Agriculture	19.8	8.5	54.3	44(18)	499	255	244	44(18)
Industry	9.3	5.2	47.6	10(4)	204	98	106	10(4)
Landfill leachate	1.7	0.29	66.1	11(3)	5.9	0.74	5.2	8(3)

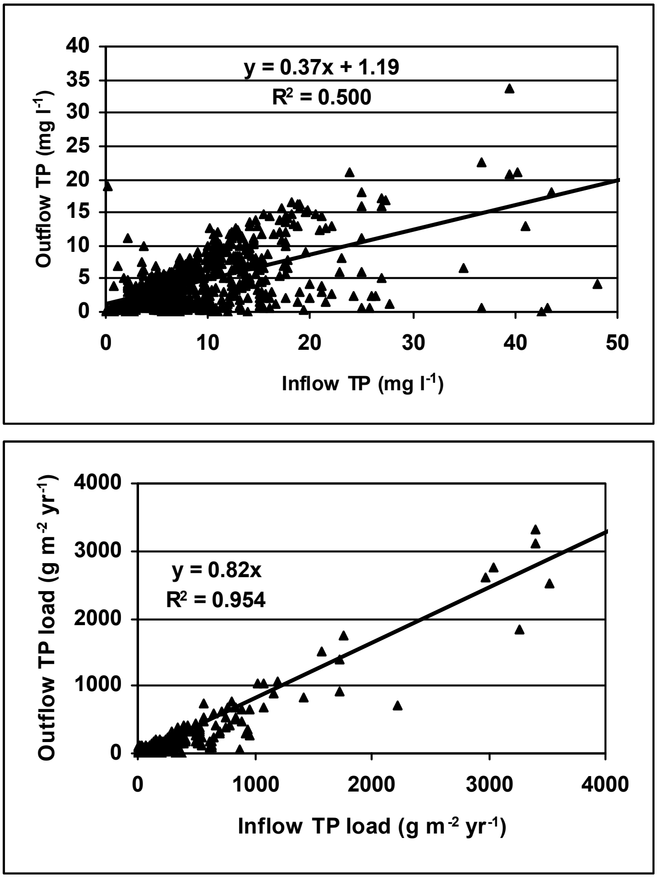


Figure 5-82. Relationship between inflow and outflow TP concentrations (top, n= 714, inflow concentration range 0.07 – 126 $mg\ l^{-1}$, highest 6 values not shown) and TP loads (bottom, n=580, inflow load range 1-7,743 $g\ m^{-2}\ yr^{-1}$, highest 4 values not shown) in HF constructed wetlands.

Rustige et al. (2003) reported based on a results from 25 HF constructed wetlands that best removal efficiency was observed at hydraulic loading rates of 1 cm d^{-1} or less with 50% of all investigates HF wetlands provided P outflow concentrations less than 2.1 mg l^{-1}. The average hydraulic loading rate in our database was 11.4 cm d^{-1}.

The inflow/outflow relationship for concentrations is stronger than those reported by Brix (1998) for 90 Danish systems:

$$C_{out} = 0.46\ C_{in} + 1.4\ (R^2 = 0.42) \tag{5.20}$$

or Vymazal (1999a) for 15 Czech systems :

$$C_{out} = 0.26\ C_{in} + 1.52\ (R^2 = 0.23) \tag{5.21}$$

but less strong than the relationship reported by Brix (1994b) for 67 systems in Denmark and United Kingdom:

$$C_{out} = 0.68\ C_{in} + 0.42\ (R^2 = 0.74) \tag{5.22}$$

Relationship between inflow and outflow loads (Fig. 5-82) is very strong but Brix (1994b) reported a similar R^2 value (0.92) for 50 Danish and UK systems.

As both aboveground biomass and P concentration in the aboveground plant tissues are similar in natural stands and constructed wetlands it is obvious that also P standing stocks in constructed wetlands should be within the range found in natural stands. Reddy and DeBusk (1987), based on literature survey, suggested the P standing stock between 1.4 and 37.5 g P m^{-2} yr^{-1} for *Typha, Phragmites, Scirpus* and *Juncus*. However, the values included also belowground standing stock which is generally not available for harvest and the authors indicated that usually >50% of the stock is stored in belowground biomass. Brix and Schierup (1989c) suggested that standing stock in aboveground biomass of emergent macrophytes, and thus available for harvesting, is roughly between 3 and 15 g P m^{-2} yr^{-1}. Vymazal (1995a) reported aboveground P standing stock in the range of 0.1 to 11 g P m^{-2} yr^{-1} for 29 various emergent species.

Vymazal (2004a) reported that P standing stock in aboveground biomass of *Typha glauca*, *Typha* spp., *Phalaris arundinacea* and *Phalaris arundinacea* growing in 30 constructed wetlands in Australia, Austria Canada, Czech Republic, Germany, Poland, The Netherlands, United Kingdom and USA varied between 0.2 and 10.5 g P m^{-2}. In Table 5-28, P standing stocks in aboveground biomass of plants growing in HF constructed wetlands is presented. The comparison with inflow P loading clearly indicate that standing stock represents only very small portion of the inflow loading.

Kuusemets et al. (2002) reported that plants (*Phragmites australis* and *Scirpus sylvaticus*) growing in a HF constructed wetland at Kodijärve, Estonia, assimilated on average 2.1 g P m^{-2} yr^{-1} with 1.15 g P m^{-2} yr^{-1} being allocated belowground. Tanner (2001) found mean combined above (including standing dead) and below-ground *Scirpus tabernaemontani* phosphorus accumulation between 8.8 and 13.4 g P m^{-2} increasing with wetland wastewater loading.

Table 5-28. Examples of phosphorus standing stock (SS, g P m^{-2}) in aboveground biomass of plants growing in HF constructed wetlands and comparison with inflow total phosphorus loading (g P m^{-2} yr^{-1}).

Plant species	Location	SS	SS as % of inflow load	Reference
Baumea articulata	Australia	5.5	2.2*	1
Phragmites australis	Poland	5.0	0.5	7
Carex fascicularis	Australia	4.6	1.8*	1
Phragmites australis	Czech Republic	3.6	1.2	5
Phragmites australis	Australia	3.4	17.8	4
Phragmites australis	Australia	3.2	24.1	4
Phragmites australis	Poland	2.8	0.25	7
Phalaris arundinacea	Czech Republic	1.5	3.7	5
Phragmites australis	Austria	2.4	2.6	6
Phragmites australis *Scirpus sylvaticus*	Estonia**	2.1	5.0	2
Phylidrum lanuginosum	Australia	0.9	0.35*	1
Phragmites australis	USA	0.7	27	3

1-Browning and Greenway (2003) *filtrable reactive P, 2- Kuusemets et al. (2002) **includes aboveground, belowground and litter, 3-Peverly et al. (1993), landfill leachate, 4-Headley et al. (2002), 5-Vymazal (1999a), 6-Haberl and Perfler (1990), 7-Obarska-Pempkowiak (1999)

For phosphorus, the same conditions as for nitrogen must be taken into consideration when the amount of P in the biomass is evaluated. Okurut (2001) observed the nitrogen uptake rate during the exponential growth of *Phragmites mauritianus* in HF system in Uganda to be 0.026 g P m^{-2} d^{-1} but during the steady state the uptake rate dropped to only 0.003 g P m^{-2} d^{-1}. Also Headley et al. (2005) noted in their study in Australia that phosphorus uptake varies greatly during the year. Langergraber (2005) in his review found daily uptake rates among wetland plants used in constructed wetlands between 0.02 and 104 mg P m^{-2} d^{-1}. Therefore, it is clear that the calculations based on short-term uptake are difficult to apply for the whole season and often overestimate the amount of phosphorus and other nutrients which could be removed via harvesting. This is especially true when only initial growth phase is considered. There is only one way how to correctly evaluate the amount of phosphorus removable via harvesting – to harvest the plants and measure biomass and P concentrations in the plant tissues.

Harvesting in temperate and cold climates usually does not result in any substantial increase in removed phosphorus. Braxton (1981) used three harvests of *Phalaris* to remove 5.4 g P m^{-2}. The first harvest in May removed 61% of the total, the second in late June 27%, and the third in late September removed additional 12%. The decrease amounts were result of decreasing respective biomass of 652 g m^{-2}, 330 g m^{-2} and 119 g m^{-2}. Suzuki et al. (1989) found for *Phragmites* that two harvests increased the yearly total biomass production by 14% over that of one harvest and increase P removal by 10%. However, harvesting *Phragmites* during the growing season may lead to serious damage of the stand because this plant translocates reserve products only very late in the season. Also, in temperate and cold climates vegetation plays an important role as insulation of wetland surface during winter and the harvest during the growing season is questionable. The situation is different under tropic climatic conditions. For example, Okurut (2001) reported that P uptake by *Cyperus papyrus* and *Phragmites mauritianus* growing in a constructed wetland in Uganda accounted for 33% and 61% of the total P removed, respectively. This was achieved by keeping the plants in exponential growth phase by regular harvesting up to four times a year. For more information on the plant effect on phosphorus removal, see Section 5.4.7.

The k_A values for various types of wastewaters our database were: 0.065 m d^{-1} (all types of wastewater), 0.035 m d^{-1} (municipal), 0.429 m d^{-1} (agricultural), 0.057 m d^{-1} (industrial) and 0.014 m d^{-1} (landfill leachate). Kadlec and Knight (1996) reported the k_A value from the North American database for TP 0.033 m d^{-1} and Brix (1998) reported the value of 0.0247 m d^{-1} for Danish systems. Kröpfelová and Vymazal (unpubl.) found an average k_A values for the Czech HF systems 0.026 m d^{-1}.

5.4.5 Microbial pollution

Wetlands are known to act as excellent biofilters through a complex of physical, chemical and biological factors which all participate in the reduction of the number of bacteria of antropogenic origin. Physical factors include mechanical filtration, straining, adsorption and sedimentation (e.g., Kadlec and Knight, 1996; Stevik et al., 2004). Chemical factors include oxidation, UV radiation, exposure to biocides excreted by some plants and adsorption to organic matter (e.g., Seidel, 1976; Ford, 1993; Vincent et al., 1994; Neori et al., 2000) Biological removal factors include antibiosis, predation by nematodes, protozoa and zooplankton, attack by lytic bacteria and viruses and natural die-off (e.g., Green et al., 1997; Decamp and Warren, 1998; Decamp et al., 1999).

Stevik et al. (2004) in their comprehensive review pointed out that two mechanisms responsible for immobilization of pathogens in wastewater moving through a porous media are straining and adsorption. The straining mechanism involves the physical blocking of movement through pores smaller that the bacteria (Griffin and Quail, 1968; Pekdeger and Matthess, 1983; Corapcioglu and Haridas, 1984; Ginn et al., 2002). Factors that influence straining are grain size of the porous media (e.g., Fontes et al., 1991; Stevik et al., 1999; Ausland et al., 2002), bacterial cell size and shape (Bitton et al., 1974; Lawrence and Hendry, 1996), degree of water saturation (Thomas and Philips, 1979; Smith et al., 1985) and clogging (e.g., Baars, 1957; Kristiansen, 1981).

Stevik et al. (2004) summarized that generally, straining becomes an important removal mechanism when the average cell size of the bacteria is greater than the size of 5% of the grains that compose the porous material (Updegraff, 1983). Bouwer (1984) reported that straining happened when the diameter of the suspended particles was larger than 0.2 times the diameter of particles constituting the porous media. There is much experimental evidence that removal of bacteria is more efficient in clogged filtration systems compared with unclogged systems (Baars, 1957; Butler et al., 1954; Krone et al., 1958; Kristiansen, 1981). However, clogging often hampers treatment efficiency of other pollutants. In porous media where the pores are larger than the bacteria, the dominant mechanism for retention of bacteria is adsorption (e.g., Gerba et al., 1975; Lance, 1984; Gerba and Bitton, 1984; Sharma et al., 1985; McDowell-Boyer et al., 1986; Harvey and Garabedian, 1991; Tan et al., 1992; Vega et al., 2003).

Solar radiation plays a very important role in the reduction of enteric bacteria (e.g., Nasim and James, 1978; Jagger, 1983). However, in HF constructed wetlands where the wastewater flows under the wetland surface UV radiation plays a minor role in inactivation process. The more important chemical process involved in bacteria inactivation in HF constructed wetlands is the excretion of various antibacterial substances by macrophyte roots (Seidel, 1976; Gopal and Goel, 1993; Vincent et al., 1994; Neori et al., 2000). According to Gopal and Goel (1993), the substances excreted by the roots of many species of wetland macrophytes responsible for this antimicrobial activity are tannic and gallic acids.

Bacteria in a biological treatment systems are subject to predation, and exposed to inhibitory substances secreted by other bacteria, bacterial phages, protozoa and nematodes (Clarholm, 1981; Peterson and Ward, 1989; Gammack et al., 1992; Mawdsley et al., 1995). Decamp and Warren (1998) and Decamp et al. (1999) pointed out that bacteria of fecal origin are grazed by a wide range of ciliate commonly found in constructed wetlands used for wastewater treatment. Many of these ciliates are filter-feeders, grazing primarily dispersed, unattached bacteria although crawling forms, such as oxytrichids, may also feed on surface-attached bacteria.

Stevik et al. (2004) summarized that the survival of intestinal bacteria differs between species (Rudolfs et al., 1950). Several *Salmonella* and *Yersinia* species have longer survival times than *Escherichia coli*, while *Shigella* and *Campylobacter* die more rapidly. Some Streptococcus species have much longer survival times than others (Stenstrøm and Hoffner, 1982). Some bacteria may survive better due to their ability to compete for nutrients with indigenous microorganisms, or they may be less susceptible to antibiotics produced by soil bacteria (Yates and Yates, 1988). Bacteria that can produce spores are more capable of surviving under unfavorable conditions than those that cannot (Acea et al., 1988).

Wastewaters contain various pathogenic or potentially pathogenic microorganisms which, depending on species concentration, pose a potential risk to human health and whose presence must therefore be reduced in the course of wastewater treatment (Hagendorf et al., 2000). Measurement of human pathogenic organisms in untreated and treated wastewater is expensive and technically challenging. Consequently, environmental engineers have sought indicator organisms that are 1) easy to monitor and 2) correlate with population of pathogenic organisms. No perfect indicators have been found, but the coliform bacteria group has been long used as the first choice among indicator organisms (Dufour, 1977). Coliforms are usually monitored as total or fecal coliforms. The fecal streptococcus (FS) group is also used frequently to confirm fecal contamination. Coliform bacteria and fecal streptococci are excreted as fecal constituents. Total coliforms (TC), however, are ubiquitous in surface waters, and they include many bacteria from the family *Enerobacteriaceae* that are not derived from human or other animal pollution sources. Contrary to FS the detection of coliforms in wastewater indicates only a possible contamination by feces, as these organisms are capable not only of surviving but also in some cases, of multiplying in water and on soil particles and plants. Thus, the total coliform measurement is the least specific indicator for providing evidence of human fecal contamination. The fecal coliform (FC or thermo-tolerant coliforms) group is composed largely of fecally derived coliforms (mostly genera *Escherichia*, *Klebsiella*, *Citrobacter, Enterobacter*), but it also includes free-living bacteria and bacteria from other warm-blooded animals including birds and mammals. Thus, although the fecal coliform measure is a better indicator of human fecal contamination than total coliform, it is by no means specific (Dufour, 1977; Kadlec and Knight, 1996).

Vymazal (2005d) in his review pointed out that the *Escherichia coli* test can be used to diagnose fecal contamination. However, because these bacteria also originate in other warm-blooded animals, the *E. coli* is not diagnostic of human fecal contamination alone. Typically, *E. coli* constitutes about 20 to 30 percent of the total coliforms found in raw and treated domestic wastewater (Dufour, 1977). The detection of FS in wastewater indicates a direct fecal contamination of the water, as FS are not, or not significantly, capable of multiplying following excretion by humans. Because fecal streptococci bacteria are more resistant to environmental stress (e.g., temperature, chemical agents) than fecal coliforms, they are used as a second indicator of fecal contamination and may also be a better indicator of the presence of the longer-living viruses originating in wastewater. However, it has also been reported that despite higher resistance towards certain environmental stress factors FS usually survive in water shorter period than bacteria of the family *Enerobacteriaceae.* Therefore, FS are considered as indicators of "fresh" pollution (Litsky et al., 1953). Also, the ratio between FC and FS is used to distinguish between human and non-human coliform contamination. Since animal feces have a higher density of FS, the FC to FS ration for non-human waste is typically less than 0.7 and the ratio for human waste is typically greater than 4.0 (Litsky et al., 1953). Because bacterial die-off affects the ratio, it is only applicable to fecal pollution within 24 h of discharge. Both FC and FS are quite easy to analyze. However, they do not necessarily correlate at all with the presence or behavior of the pathogens in human feces.

In Table 5-29, analysis of removal of enteric bacteria in HF constructed wetlands is shown. Removal varies among indicators between 1 and 3 $\log_{10}$ with best performance recorded for fecal coliforms (3.05 $\log_{10}$ CFU 100 ml^{-1}). However, the outflow numbers of enteric bacteria are usually still too high for agricultural reuse.

Table 5-29. Removal of indicator enteric bacteria in HF constructed wetlands. Data from systems treating municipal and agricultural wastewaters. *n= number of entries (annual means) with number of systems in parentheses.

	CFU 100 ml^{-1}		**Eff. (%)**	**n***	**Log_{10} CFU 100 ml^{-1}**		
	In	**Out**			**In**	**Out**	**Rem**
Fecal coliforms	1.67×10^8	1.47×10^5	92.4	141(95)	8.22	5.17	3.05
Total coliforms	1.52×10^9	1.78×10^8	90.5	75(52)	9.18	8.25	0.93
Fecal streptococci	7.58×10^5	2.89×10^4	90.8	36(26)	5.88	4.46	1.42
Escherichia coli	3.61×10^8	7.23×10^7	90.1	48(26)	8.59	7.86	0.76

García et al. (2003) reported that removal of microbial indicators (fecal coliforms and somatic coliphages) in HF constructed wetlands depends on hydraulic retention time and size fraction of the filtration medium. The authors found in their study at Montcada i Reixac near Barcelona, Spain, that there is a positive relationship between microbial inactivation and HRT, but in general when the HRT is approximately 3 days the microbial inactivation reaches saturation values. The authors concluded that this trend is possibly related to the fact that the assumptions of the first-order microbial decay is not valid as the HRT rises.

The microbial inactivation of FC ranged between 0.1 and 2.7 log-units in beds with coarser granular material (5-25 mm) and between 0.7 and 3.4 log-units in the bed with finer material (2-13 mm) depending on the HRT. The authors also pointed out that it seems difficult to remove more than 3 log-units of fecal coliforms and somatic coliphages with reed beds similar to those used in their experiments. They concluded that the specific surface area necessary to achieve this removal is approximately 3 m^2 PE^{-1} (García et al., 1998).

Also Tanner et al. (1998a) observed a clear dependence of FC removal on HRT. In their study the authors observed the increase of FC removal from 1.13 to 2.43 log-units for HRT 2 and 9.5 days, respectively. Contrary to findings of García et al. (1998) the authors found that FC removal rates were monotonically related to loading rates, and could be modeled using a simple plug-flow, first-order approach accounting for removal down to non-zero background concentrations.

Verlicchi et al. (2004) studied the combination of low dosage of sodium hypochlorite (NaClO) and HF constructed wetland to disinfect secondary treated municipal wastewater. The experiments were carried out at the wastewater treatment plant in Ferrara, northern Italy (240 000 PE). The investigation have shown that the removal of *Escherichia coli* amounted to 5 log units using a combination of low dosage of NaClO (2 mg l^{-1}) and a HF constructed wetland with a specific area of 0.54 m^2 PE^{-1}. The final effluent met the standards for unrestricted reuse in agriculture with concentrations of undesirable and toxic by-products, particularly trihalomethanes, trichloroethylene and tetrachloroethylene well below the limits of the Italian Regulations for water reuse.

Butler et al. (1993) suggested that HF constructed wetlands are able to remove both bacteria and viruses from sewage with exponential decay with respect to distance along the channel. The linear regression form of the exponential equation for fecal coliform removal in HF is:

$$\log_{10} (\text{CFU ml}^{-1}) = 4.04 - 0.02 \text{ (bed length in meters)} \tag{5.23}$$

Williams et al. (1994) suggested the outflow fecal coliform number according to the following equation:

$$\log_{10} (\text{CFU ml}^{-1}) = 0.074 \text{ x } BOD_5 + 1.95 \tag{5.24}$$

Decamp and Warren (2000) and Warren et al. (2000) reported that the presence of *Phragmites australis* also appears to have a beneficial effect on *Escherichia coli* removal. The results indicate that the length of the planted gravel bed required to give a ten-fold reduction in counts of *E. coli* in the unplanted system was at least twice that required for the equivalent reduction in the planted gravel bed. The authors pointed out that higher bacteria removal in planted wetlands might be caused by the root exudates which are toxic to a range of bacteria. The similar results were reported by Rivera et al. (1995) from Mexico. The removal of fecal coliform bacteria at San Luis Potosí constructed wetland amounted to 63-74% in cells planted with *Typha* and *Phragmites* while in soil without reeds the removal was only 16%. Hench et al. (2003) reported higher FC removal in wetlands planted with *Typha latifolia*, *Juncus effusus* and *Scirpus validus* as compared to unvegetated filters. Gersberg et al. (1987b) found significantly higher removal of coliform bacteria in wetland cells in Santee, California, planted with *Scirpus validus* (Bulrush) (99.1%) as compared to unplanted wetland cells (95.7%). Positive effect on bacteria removal in planted wetlands as compared to unplanted ones was reported by Gersberg et al. (1989b), Vacca et al. (2005) or Soto et al. (1999).

Thurston et al. (2001) reported average reduction of coliphage 95.2% (average ouflow 4.7 PFU ml^{-1}) which was comparable with results reported by Gersberg et al. (1987a,b) who reported reduction of 98.3% for MS-2 bacteriophage in wetland cells planted with *Scirpus validus* in Santee, California.

Constructed wetlands are also very effective in removal of protozoan parasites such as *Cryptosporidium parvum* or *Girardia lamblia* (Stott et al., 2001; Hill, 2003). The removal mechanisms, however, remain unknown. Stott et al. (2001) reported that protozoan (e.g., *Paramecium caudatum*, *Stylonychia mytilus*, *Euplotes patella*) predation may be an important factor in the removal of *Cryptosporidium* oocysts from wastewaters in constructed wetlands. *Paramecium caudatum* exhibited the highest grazing rate. This ciliate was identified in several HF constructed wetlands (Vymazal et al., 2001) and it is probably a very common species in HF constructed wetlands.

Thurston et al. (2001) reported that *Girardia* cysts and *Cryptosporidium* oocysts were reduced by an average of 87.8% and 64.2%, respectively in an experimental facility in Arizona. The respective numbers in 100 l were 0.7 and 2.7. The authors pointed out that *Girardia* cysts may be removed more effectively because of their larger size. Also, the authors stressed that the detection methods for both *Girardia* and *Cryptosporidium* were not a viability test and, therefore, the removal of infectious organisms could be greater than indicated. Nokes et al. (2003) reported *Giradia* and *Cryptosporidium* removals > 97.8% and > 44.3% to > 77.1%, respectively in two oin-site HF constructed wetlands in Arizona. Quiňónez-Diaz et al. (2001) reported removal of *Girardia* cysts and *Cryptosporidium* oocysts > 99.9% and >99%, respectively in a subsurface experimental wetland in Arizona planted with *Scirpus olneyi*. Removal of viruses and parasites in HF constructed wetlands has also been discussed by Stott et al. (2002) based on the results from United Kingdom, Mexico and Egypt.

5.4.6 Heavy metals

Transformations and retention mechanisms of heavy metals in constructed wetlands are manifold (see Section 2.7). In general, FWS constructed wetlands are more commonly used to treat waters with high heavy metal concentrations (see Section 4.1.4.5) because oxic processes are more effective. HF systems which are mostly anoxic/anaerobic provide less opportunity for heavy metal retention. Probably, the most extensive process is formation of insoluble metal sulfides. In the literature, there are several models to predict metal retention (e.g. Baker et al., 1991; Flanagan et al., 1994; Mitsch and Wise, 1998) but most models deal with iron retention in FWS wetlands. Mlay et al. (2006) modelled retention in copper in HF wetlands treating acid mine drainage in Tanzania. The authors concluded that regression analyses indicated a good correlation between the simulated and observed values for inlet/outlet. However, they also pointed out that still there are many factors which were not taken into consideration because these are difficult to estimate.

In the literature, only limited information exists on the removal of heavy metals in HF constructed wetlands. Pantano et al. (2000) reported the use of HF constructed wetlands to treat mining impacted groundwater in Montana. The authors monitored gravel based unit and a unit with gravel with 20% (v/v) compost substrate. The system was very effective in removal of cadmium, copper and zinc while arsenic concentrations increased in the outflow. Lead concentration remained unchanged after the passage through the wetland.

Byekwaso et al. (2002) monitored a FWS-HF constructed wetland treating an effluent from Kasese Cobalt Company's cobalt recovery processing plant. A 1000 m^2 HF wetland filled with limestone and planted with *Phragmites mauritianus* received water from a 800 m^2 FWS wetland. The system efficiently removed Pb, Co, Ni, Cu, Cd and Fe. There was no overall removal of zinc and manganese was released from the system. While zinc was released in FWS part and removed in HF part, manganese was released in both FWS and HF parts.

Lim et al. (2003) reported Zn, Pb and Cd removal efficiency in a small outdoor HF units in Malaysia of over 99%. When treating the metals in combination, the sorption capacity of the gravel media was strongly reduced due to competition effect among the metals.

Vymazal and Krása (2005) observed that retention of metals (Al, Cd, Cu, Fe, Mn, Ni, Pb, Zn) in a HF constructed wetland treating municipal sewage varied widely among elements. On average, 11% of the inflow load of metals was retained in the aboveground biomass of *Phalaris arundinacea* nd *Phragmites australis* while 71% of the inflow load was retained in the sediment and/or in the belowground biomass.

Vymazal (2005e) pointed out that substantial amounts of heavy metals and aluminum are retained in pretreatment units in the Czech Republic constructed wetland systems. Similar observations were reported by Lesage (2006) in Belgium.

Ranieri (2004) has monitored a large HF constructed wetland (two beds 65 x 30 m) in Putignano, southern Italy for 48 months. The wastewater is taken from the Putignano treatment plant which treats mainly municipal wastewater with a percentage of electroplating and textile industries and stormwater runoff with elevated concentrations of Ni and Cr. The balance for Ni and Cr revealed that over the 48-month period the most metals were bound in sediments (90% Cr and 73% Ni as compared to inflow load) while in plant tissues only 4% Cr and 3% Ni were sequestered.

In Tables 5-30 to 5-40, removal of eleven heavy metals and several risk elements is presented. Most elements presented in this survey are retained but to various extents. On the other hand, iron (Table 5-35) and especially manganese (Table 5-37) and arsenic (Table 5-31) are often released from the systems as a consequence of reducing conditions (see section 2.7.1).

Table 5-30. Removal of aluminum in HF constructed wetlands.

Locality	Concentration ($\mu g\ l^{-1}$)		Loading ($mg\ m^{-2}\ d^{-1}$)			Reference
	In	Out	In	Out	Rem.	
Nučice, Czech Republic	451	<40	12	<1.1	>10.9	Vymazal and Krása (2003)
Břehov, Czech Republic	1,290	605	88	41	47	Unpubl. results
Slavošovice, Czech Republic	398	113	21	6	15	Unpubl. results
Monroe Co., NY	80	<50	0.23	<0.14	>0.09	Eckhardt et al. (1999)
Mořina, Czech R.	551	<40	23	<1.7	>21.3	Vymazal (2005e)
Zemst-Kesterbeek, Belgium	1,218	86	98	6.9	91.1	Lesage (2006)

Table 5-31. Removal of arsenic in HF constructed wetlands.

Locality	Concentration ($\mu g\ l^{-1}$)		Loading ($mg\ m^{-2}\ d^{-1}$)			Reference
	In	Out	In	Out	Rem.	
Břehov, Czech Republic	0.8	6.5	0.05	0.44		Unpubl. results
Slavošovice, Czech Rep.	5.5	2.2	0.30	0.12	0.18	Unpubl. results
Zemst-Kesterbeek, Belgium	0.71	5.6	0.06	0.45		Lesage (2006)
Hasselt-Kiewit, Belgium	3.0	11	0.08	0.29		Lesage (2006)

Table 5-32. Removal of cadmium in HF constructed wetlands. *amended secondary activated sludge effluent.

Locality	Concentration ($\mu g\ l^{-1}$)		Loading ($\mu g\ m^{-2}\ d^{-1}$)			Reference
	In	Out	In	Out	Rem.	
Przywidz, Poland	3.0	1.48	450	220	230	Obarska-Pempkowiak (2000, 2003)
Nučice, Czech Rep.	0.43	0.10	11	2.6	8.4	Vymazal (2003b)
Břehov, Czech Rep.	0.28	0.27	19	18	1	Unpubl. results
Santee, California*	532	<4.0	25,000	188	24,812	Gersberg et al. (1984)

Table 5-33. Removal of chromium in HF constructed wetlands.

Locality	Concentration ($\mu g\ l^{-1}$)		Loading ($\mu g\ m^{-2}\ d^{-1}$)			Reference
	In	Out	In	Out	Rem.	
Monroe Co., NY	9.0	<5.0	26	<14	>12	Eckhardt et al. (1999)
Zemst-Kesterbeek, Belgium	2.1	0.95	0.17	0.08	0.09	Lesage (2006)
Hasselt-Kiewit, Belgium	5.4	0.32	0.15	0.01	0.14	Lesage (2006)
Putignano, Italy	84	13.5	3,150	510	2,640	Ranieri (2004)

Table 5-34. Removal of copper in HF constructed wetlands. *amended secondary activated sludge effluent.

Locality	Concentration ($\mu g\ l^{-1}$)		Loading ($mg\ m^{-2}\ d^{-1}$)			Reference
	In	Out	In	Out	Rem.	
Przywidz, Poland	18.2	7.6	2.73	1.15	1.58	Obarska-Pempkowiak (2000, 2003)
China, pig farm	4,700	200	860	37	823	Junsan et al. (2000)
Nučice, Czech Republic	11.3	<2.0	0.30	<0.05	>0.25	Vymazal and Krása (2003)
Mořina, Czech Republic	4.4	<2.0	0.19	<0.08	>0.11	Vymazal (2005e)
Zemst-Kesterbeek, Belgium	104	0.34	8.4	0.03	8.37	Lesage (2006)
Hasselt-Kiewit, Belgium	78	6.1	2.1	0.17	1.93	Lesage (2006)
Santee, California*	9,260	56	741	5	736	Gersberg et al. (1984)
Santee, California*	10,680	59	428	2	426	Gersberg et al. (1984)

Table 5-35. Removal of iron in HF constructed wetlands.

Locality	Concentration ($\mu g\ l^{-1}$)		Loading ($mg\ m^{-2}\ d^{-1}$)			Reference
	In	Out	In	Out	Rem.	
Nučice, Czech Republic	1,096	721	29	19	10	Vymazal (2003b)
Břehov, Czech Republic	1,490	1,638	102	112		Unpubl. results
Slavošovice, Czech Republic	950	2,425	51	130		Unpubl. results
Monroe Co., NY	15,000	1,300	43	3.7	39.3	Eckhardt et al. (1999)
Esval, Norway	9,900	9,100	1,414	1,300	114	Mæhlum et al. (1999)
Bølstad, Norway	6,720	1,344	336	67	269	Mæhlum et al. (1999)
Mořina, Czech Republic	301	348	12.5	14.4		Vymazal (2005e)
Zemst-Kesterbeek, Belgium	718	1,224	58	99		Lesage (2006)

Table 5-36. Removal of lead in HF constructed wetlands.

Locality	Concentration ($\mu g\ l^{-1}$)		Loading ($mg\ m^{-2}\ d^{-1}$)			Reference
	In	Out	In	Out	Rem.	
Przywidz, Poland	15.7	7.3	2.35	1.1	1.25	Obarska-Pempkowiak (2000, 2003)
Nučice, Czech Republic	120	2.8	3.2	0.08	3.12	Vymazal (2003b)
Mořina, Czech Republic	5.84	<2.0	0.25	<0.08	>0.17	Vymazal (2005e)
Zemst-Kesterbeek, Belgium	8.7	ND	0.7	ND	0.7	Lesage (2006)
Hasselt-Kiewit, Belgium	47	3.1	1.25	0.08	1.17	Lesage (2006)

Table 5-37. Removal of manganese in HF constructed wetlands.

Locality	Concentration ($\mu g\ l^{-1}$)		Loading ($mg\ m^{-2}\ d^{-1}$)			Reference
	In	Out	In	Out	Rem.	
Nučice, Czech Republic	278	53	7.4	1.4	6.0	Vymazal and Krása (2003)
Mořina, Czech Republic	109	402	4.5	16.7		Vymazal (2005e)
Břehov, Czech Republic	143	290	9.8	19.9		Unpubl. results
Slavošovice, Czech Republic	147	41	7.9	2.2	5.7	Unpubl. results
Zemst-Kesterbeek, Belgium	79	169	6.4	13.7		Lesage (2006)

Table 5-38. Removal of mercury in HF constructed wetlands.

Locality	Concentration ($\mu g\ l^{-1}$)		Loading ($\mu g\ m^{-2}\ d^{-1}$)			Reference
	In	Out	In	Out	Rem.	
Břehov, Czech Republic	0.017	0.006	1.2	0.4	0.8	Unpubl. results
Slavošovice, Czech Republic	0.024	0.023	1.3	1.2	0.1	Unpubl. results
Zemst-Kesterbeek, Belgium	0.038	0.006	3.1	0.5	2.6	Lesage (2006)
Hasselt-Kiewit, Belgium	0.26	0.035	7.0	0.9	6.1	Lesage (2006)

Table 5-39. Removal of nickel in HF constructed wetlands.

Locality	Concentration ($\mu g\ l^{-1}$)		Loading ($mg\ m^{-2}\ d^{-1}$)			Reference
	In	Out	In	Out	Rem.	
Nučice, Czech Republic	9.5	0.7	0.25	0.02	0.23	**Vymazal (2003b)**
Monroe Co., NY	60	30	0.17	0.085	0.085	Eckhardt et al. (1999)
Mořina, Czech R.	1.67	4.61	0.07	0.19		Vymazal (2005e)
Zemst-Kesterbeek, Belgium	2.8	0.54	0.23	0.04	0.19	Lesage (2006)
Hasselt-Kiewit, Belgium	3.8	ND	0.1	ND	0.1	Lesage (2006)
Putignano, Italy	52	12.7	1.95	0.47	1.48	Ranieri (2004)

Table 5-40. Removal of zinc in HF constructed wetlands. *amended secondary activated sludge effluent.

Locality	Concentration ($\mu g\ l^{-1}$)		Loading ($mg\ m^{-2}\ d^{-1}$)			Reference
	In	Out	In	Out	Rem.	
Nučice, Czech Republic	198	<5	5.3	<0.13	>5.17	Vymazal and Krása (2003)
Břehov, Czech Repblic	280	37	19.2	2.5	16.7	Unpubl. results
Slavošovice, Czech R.	98	46	5.3	2.5	2.8	Unpubl. results
Monroe Co., NY	80	25	0.23	0.07	0.16	Eckhardt et al. (1999)
Esval, Norway	100	61	18.6	11.3	7.3	Mæhlum et al. (1999)
Mořina, Czech Republic	73	6.4	3.0	0.3	2.7	Vymazal (2005e)
Zemst-Kesterbeek, Belgium	130	3.0	10.5	0.2	10.3	Lesage (2006)
Hasselt-Kiewit, Belgium	574	58	15.4	1.6	13.8	Lesage (2006)
Santee, California*	11,280	370	530	17	513	Gersberg et al. (1984)

5.4.7 The influence of vegetation

Dense beds of emergent wetland plants are the most obvious visual feature of HF constructed wetlands. They undoubtedly play a major role in enhancing their wildlife habitat values, aesthetics and perceived naturalness (Tanner, 2001). It is obvious that plants provide many benefits to the treatment process but their role in overall treatment performance is still not unanimous (see Section 3.5). Tanner (2001) in his review on the role of plants in HF constructed wetlands identified organic matter production and root-zone oxygen release as key factors influencing nutrient transformation and sequestration.

It is generally assumed that planted wetlands outperform unplanted controls mainly because the plant rhizosphere stimulates microbial community Gagnon et al. (2006). It has been suggested that plant rhizosphere enhances microbial density and activity by providing root surface for microbial growth, a source of carbon compounds through root exudates and a micro-aerobic environment via root oxygen release (e.g., Gersberg et al., 1986). Higher microbial densities in planted systems were reported in full scale constructed wetlands with 2.7 x 10^6 CFU g^{-1} of gravel evaluated by culture method (Hatano et al., 1993) and in planted microcosms with 3.2 x 10^9 g^{-1} of gravel measured by fluorescence microscopy (Münch et al., 2005). Collins et al. (2004) showed that, while microbial populations will grow on any surface, plants influence microbial composition and abundance. Several authors have shown plants differ in root surface area available for bacterial growth (Hatano et al., 1993; Vymazal et al., 2003, Kyambadde et al., 2004; Gagnon et al., 2006).

Tanner et al. (1995a) reported that removal of BOD_5, suspended solids and fecal coliforms was not affected by the presence of vegetation in experimental wetlands planted with *Scirpus validus* in Ruakura. New Zealand. However, Tanner et al. (1995b) reported substantial increase in removal of both nitrogen and phosphorus in the same wetland (Table 5-41).

Table 5-41. Removal of total nitrogen and total phosphorus in planted and unplanted constructed wetlands ar Ruakura, New Zealand at four various hydraulic retention times. Data from Tanner et al. (1995b).

HRT (d)	**Removed TN load (g m^{-2} d^{-1})**		**Removed TP load (g m^{-2} d^{-1})**	
	Planted	**Unplanted**	**Planted**	**Unplanted**
2	1.3 (48)	0.4 (12)	0.32 (37)	0.08 (12)
3	0.85 (52)	0.8 (39)	0.20 (45)	0.22 (41)
5.5	0.77 (71)	0.32 (25)	0.20 (67)	0.11 (35)
7	0.50 (75)	0.30 (41)	0.13 (74)	0.07 (36)

Coleman et al. (2001) found in experimental units that depth of gravel (45 or 60 cm) had little effect on wetland treatment ability, but did influence

Typha latifolia and *Scirpus cyperinus* growth patterns. Gravel alone provided substantial wastewater treatment, but vegetation further improved many treatment efficiencies. *Typha* sugnificantly outperformed *Juncus effusus* and *S. cyperinus* in effluent quality improvement. In addition, species mixture out-performed species monoclutures. *T. latifolia* was the superior competitor in mixtures.

Gersberg et al. (1983) reported only slightly better removal effect for total nitrogen removal in vegetated (27%) and unvegetated (25%) beds pointing out that plant uptake may only account for a small fraction of the overall nitrogen removal observed. However, Gersberg et al. (1986) found in the same facility at Santee, California, that the presence of plants significantly affected the removal of ammonium-N. During the period from August 1983 to December 1984, the mean influent of NH_4^+-N concentration of 24.7 mg l^{-1} was reduced to mean effluent levels of 1.5 mg l^{-1} (93.9% reduction) for the *Scirpus validus* bed, 5.4 mg l^{-1} (78.1%) for the *Phragmites australis* bed and 17.7 mg l^{-1} (28.3%) for the *Typha latifolia* bed, as compared to a mean value of 22.1 mg l^{-1} (10.5%) for the unvegetated bed. The *Scirpus* and *Phragmites* beds were superior at removing ammonia, and both produced effluent values significantly lower than did the *Typha* bed ($p < 0.01$). Finlayson and Chick (1983) found exactly the same order of efficiency for NH_4^+-N (and TKN) removal from abattoir effluent in Australia. The removal effect was 54% (56%) for *Scirpus validus,* 12% (26%) for *Phragmites australis* and 3% (14%) for a mixture of *Typha domingensis* + *T. orientalis*. For total P removal, the order was slightly different but the wetland with *Scirpus* showed the highest removal effect (61%). In this case, removal in *Typha* bed (53%) was superior to that with *Phragmites* (37%).

Kaseva (2004) reported positive effect of plants on treatment performance of a constructed wetland in Tanzania (Table 5-42), especially for removal of COD and ammonia. Also Mbuligwe (2004) observed better treatment performance of cells planted with *Typha latifolia* and *Colocasia esculenta* as compared to unplanted cell in a system treating anaerobically pretreated wastewater in Dar es Salaam, Tanzania.

Table 5-42. Comparison HF constructed wetlands planted with *Phragmites mauritianus* and *Typha latifolia* with an unplanted filter in Tanzania. Data from Kaseva (2004).

			Outflow		
Parameter	**Unit**	**Inflow**	**Unplanted**	***Phragmites***	***Typha***
NH_4-N	mg l^{-1}	20.6	18.3	15.1	15.9
NO_x-N	mg l^{-1}	1.62	1.1	0.97	0.93
COD	mg l^{-1}	106	71	46.5	41.8
FC	Log_{10} CFU 100 ml^{-1}	7.1	6.7	6.6	6.6
TC	Log_{10} CFU 100 ml^{-1}	8.4	7.2	7.0	7.0

Karathanasis et al. (2003) studied the effect of various plants on the performance of twelve HF constructed wetlands for domestic wastewater from a single house in Kentucky. The systems served two to five family members and consisted of septic tank and a single lined wetland cell of varying size (34 to 60 m^2). The treatment cells were 41-46 cm deep and filled with limestone gravel 2.5 to 6 cm in diameter. Three wetlands were planted with *Typha latifolia* (Broadleaved cattail), three with *Festuca arundinacea* (Fescue) and three with a variety of plants (polyculture) consisting mainly of *Iris pseudacorus* (Yellow flag), *Canna x. generalis* (Common garden canna), *Hemerocallis fulva* (Day lilies), *Hibiscus moscheutos* (Hibiscus), *Scirpus validus* (Soft-stem bulrush) and *Mentha spicata* (Spearmint). Three systems remained unplanted. The vegetated systems showed significantly greater ($P < 0.05$) removal efficiencies for BOD (> 75%) and TSS (> 88%) than the unplanted systems (63 and 46%, respectively) throughout the year. The findings suggested no significant difference ($P < 0.05$) in the average yearly removal of fecal bacteria between systems, with the vegetated systems performing best during warmer months and the unplanted systems performing best during the winter. Overall, the polyculture systems seemed to provide best and most consistent treatment for all wastewater parameters. This is in agreement with other authors (Karpiscak et al., 1996; Coleman et al., 2001). The Fescue systems were generally inferior to the polyculture and Cattail systems because of their shallow rooting zone and limited biofilm surface area, while the unplanted systems were completely inefficient for BOD and TSS removal (Karathanasis et al., 2003). In spite of better overall treatment of fecal coliforms provided by the vegetated systems, especially during the warmer months of the year, the yearly performance differences from the unplanted systems were rather statistically significant. There results are in agreement with those reported by Green et al. (1997) but contradict other findings in which the presence of vegetation dramatically improved FC removal (Rivera et al., 1995; Soto et al., 1999).

Akratos and Tsihrintzis (2007) found in experimental HF constructed wetlands that the presence of plants and namely *Typha latifolia* increased removal of TKN (66.8% versus 34.2%) and phosphorus (58.4% vs. 42.5%). Removal of COD was only slightly higher in planted units (89.3%) as compared to unplanted units (87.2%). Burgoon et al. (1989) reported that TKN removal was highest in gravel-based mesocosms planted with *Sagittaria latifolia* (91.8%), followed by *Typha latifolia* (85.6%), *Scirpus pungens* (75.5%) and *Phragmites australis* (67.5%). The removal of TKN in an unplanted mesocosm amounted to only 55.2%. Removal of BOD_5 was much less affected by the presence of plants – removal of BOD_5 in an unplanted mesocosms was 87.6% while removals in planted mesocosms varied between 85.2 and 93.2%. Dallas and Ho (2005) reported higher removal of BOD_5 and fecal coliforms in HF experimental wetlands planted with *Coix lacryma-jobi* as compared to unplanted units in Costa Rica.

Naylor et al. (2003) reported that during the use of experimental HF constructed wetlands to treat diluted sludge from a freshwater fish farm anaerobic digester planted wetlands (*Phragmites australis, Typha latifolia*) clearly outperforming unplanted units in term of BOD_5, COD, TKN, NH_4-N. The removal of TSS, NO_3-N PO_4-P and TP were comparable. Drizo et al. (1995) found no statistical difference in redox potential values in planted (*Phragmites australis*) and unplanted HF pilot-scale units.

Haule et al. (2002) observed substantially higher removal of ammonia nitrogen (59-92%) in experimental constructed wetlands planted with various macrophytes (*Phragmites mauritianus, Typha domingensis, T. capensis, Cyperus grandis, C. dubius* and *Kyllinga erectus*) as compared to unplanted control (44%). TKN removal in planted wetlands varied between 50-73% while unplanted control exhibited only 28% removal. El Hafiane and El Hamouri (2004) found that HF wetlands planted with *Arundo donax* removed 21-25% COD, 29-41% TSS, 35-66% NH_4-N and 14-47% TP more than unplanted units in Rabat, Morocco. Zhou et al. (2004) found in HF constructed wetlands for treatment of agricultural runoff that beds planted with *Phragmites australis* and *Zizania caduciflora* removed substantially more nitrogen than unplanted bed. At various HRTs the *Phragmites* bed removed 43-80% of TN, *Zizania* bed removed 27-79% of TN while unplanted bed removed only 33-65% of TN.

de Lucas et al. (2006) observed that the plants with high growth (*Phragmites australis, Lythrum salicaria* and *Iris pseudacorus*) improved TN and TP removal from winery wastewater and adversely affected sulfate anaerobic reduction. Pandey et al. (2006) found substantially higher removal of pollutants in HF system in Nepal planted with *Phragmites karka* (Table 5-43).

Table 5-43. Comparison of treatment performance of HF constructed wetland planted with *Phragmites karka* and unplanted unit. Values in mg l^{-1}. Data from Pandey et al. (2006).

	Inflow	**Outflow**	
		Planted	**Unplanted**
BOD_5	321	48	89
COD	654	87	122
TSS	157	61	63
NH_4-N	33.9	24	26
TKN	49.7	33	41
TP	2.99	1.25	1.99

Hook et al. (2003) pointed out that the presence of plants can strongly affect the seasonal patterns of water treatment in HF constructed wetlands. The authors observed in temperature-controlled experimental units that compared to controls, plants enhanced COD removal overall, and they either attenuated (*Typha latifolia*) or eliminated (*Carex rostrata* and *Scirpus acutus*) the seasonal decrease in performance expected at low temperatures.

5.4.8 Seasonal variation in treatment effect

The opinion of the effect of season or temperature on the HF constructed wetlands treatment performance is far from being unanimous. In the literature, there are reports suggesting little or now seasonal effects as well as reports indicating strong seasonal dependence. In reality, HF constructed wetlands are successfully operated in cold climates. For example, Giæver (2003) reported on the HF system in Fagernes, which is located in the northern part of Norway at 68° northern latitude. There are also other fine examples of the use of HF constructed wetlands in cold climate (e.g., Wallace and Knight, 2006; Navara, 1996). Comprehensive information on season, temperature and climate effects on treatment performance of HF systems can be found in Mander and Jenssen (2003).

Kadlec et al. (2003) described two adjacent HF system in Minnesota with average annual air and wetland water temperatures of 5°C and 9°C with air temperatures reaching below -40°C. The results revealed the highest removal of BOD_5 during autumn and summer with substantial decrease in treatment efficiency during winter and spring. Similar results were obtained for nitrogen and phosphorus removal, where highest mass removal occurred in summer. Also, removal of fecal coliforms was highest in summer. Removal of suspended solids was affected by season only slightly. However, this is not surprising as removal of suspended solids occurs mostly through physical factors.

On the other hand, Züst and Schönborn (2003) could not find any influence of wastewater temperature on the removal efficiency of COD, ammonia-N and TP in constructed wetlands located at the altitude of 730 m with average annual temperature of 8.4°C in Switzerland. The treatment efficiency was still very good at temperatures as low as 0.5°C. Also Mæhlum and Jenssen (2003) reported that there was no significant difference in efficiency between cold (< 4°C) and warm (>11°C) periods for all parameters tested (TP, TN, COD, BOD_7, TSS) in 9 constructed wetlands in Norway.

Hook et al. (2003) pointed out that also the presence of plants can strongly affect the seasonal patterns of water treatment in HF constructed wetlands. The authors observed in a temperature-controlled experiment (4-24°C) in Montana that performance of unplanted controls followed temperature closely and varied widely between winter and summer. *Typha latifolia* units varied less than the controls and followed temperature less closely. In contrast, performance of *Carex rostrata* and *Scirpus acutus* units was relatively constant throughout the year. At the lowest temperature, COD removal averaged approximately 60% with Typha, 85% with *Scirpus* and >90% with *Carex*. The poor performance of *Typha* compared to *Scirpus* and *Carex* has been attributed to a shallower root system (Campbell and Ogden, 1999). Hook et al. (2003) also pointed out that the excellent performance of *Carex rostrata* can be connected to its belowground structures. The proportion of belowground biomass and production is relatively high (Bernard, 1974; Konings et al., 1989; Aerts and de Calluwe, 1995), roots grow in fall and winter and belowground biomass reaches its annual maximum in winter (Bernard, 1974; Aerts and de Calluwe, 1994), roots occupy the upper 30 cm densely but extend tu up 230 cm deep (Saarinen, 1996) and roots morphology apparently favors oxygen transport.

Hill et al. (2003) reported that treatment of soluble P in a dairy farm in northern New York State, was not significantly different in summer and winter. Instead, percent removal of soluble P was determined by substrate type. In another study dealing with dairy wastewaters, Kern (2003) found no seasonal effect on the removal of COD, TN, TP and fecal coliforms. The author pointed out that temperature had a great influence on bacterial numbers, at least on proteolytic, nitrifying and fecal coliform bacteria, which are not adapted to low temperatures prevailing during the winter period (average 3.1°C). However, in contrast to bacterial numbers, bacterial activities did not reflect clear restrictions due to low temperatures. Aerobic and anaerobic consumers of carbon sources seemed to be dependent primarily on carbon supply and not on high temperatures. Only N_2O production was lower during the winter than during the summer treatment.

Vymazal et al. (2003) found no substantial difference in removal of total coliforms, fecal coliforms and fecal streptococci) during summer and winter in six HF constructed wetlands in the Czech Republic. Vymazal (2001c) reported very little difference between removal efficiency of organics in summer and winter in two HF constructed wetlands in the Czech Republic during the period 1994-1999 (Table 5-44).

In Table 5-45, seasonal results from a HF system in Korea are shown (Ham et al., 2004). The figures clearly indicate that the evaluation based on inflow and outflow concentrations may be misleading. The removal percentage is lower during winter, however, when removed loadings are compared, the winter performance is comparable with growing season with the exception of total nitrogen where concentration reduction and removed loading are both lower during winter.

Steinmann et al. (2003) did not observed any seasonal changes in BOD_5, COD and TN removal in a HF system in Mörlbach, Germany. Only the removal of ammonia-N was slightly lower during the winter period (inflow 4.4 mg l^{-1}, outflow 3.9 mg l^{-1}). However, the removal of NH_4-N was low in the summer anyway (inflow 3.2 mg l^{-1}, outflow 2.6 mg l^{-1}). Similar results were reported by Dahab and Surampalli (2001) from Nebraska, U.S.A.

Removals of BOD_5, COD and suspended solids were quite similar in summer (mean influent temperature 17.7°C) and winter (10.1°C) periods while removal of NH_4-N, NO_3-N and TP were somewhat lower during winter.

Table 5-44. Removal of BOD_5 and COD in HF constructed wetlands at Ondřejov and Koloděje, Czech Republic. "Summer" and "winter" periods: May-October and November-April, respectively. Average "summer" water temperatures 12.9°C at Ondřejov and 13.8°C at Koloděje. "Winter" average water temperatures 6.2°C at Ondřejov and 5.2°C at Koloděje. Data from Vymazal (2001c).

	BOD_5				**COD**			
	Summer		**Winter**		**Summer**		**Winter**	
	IN	**OUT**	**IN**	**OUT**	**IN**	**OUT**	**IN**	**OUT**
Koloděje	92	9.8	101	10.8	232	42	252	48
Ondřejov	96	13.1	87	10.5	215	48	247	41

Table 5-45. Performance of a HF CW at Konkuk University campus, Seoul, Korea during the period of July 1998 - December 2003. E = removal efficiency in %, Ret. = retained load. Data from Ham et al. (2004).

	Concentration (mg l^{-1})						**Loading (kg ha^{-1} d^{-1})**					
	Growing season			**Winter**			**Growing season**			**Winter**		
	In	**Out**	**E**	**In**	**Out**	**E**	**In**	**Out**	**Ret.**	**In**	**Out**	**Ret.**
BOD_5	116	21	***81***	151	62	***62***	73	13	***60***	94	39	***55***
TSS	62	14	***72***	103	65	***65***	39	9	***30***	64	21	***43***
TP	13	7	***44***	13	27	***27***	8	4	***4***	8	5	***3***
TN	122	94	***20***	120	108	***8***	76	59	***17***	75	68	***7***

Gorra et al. (2007) reported no significant differences in BOD_5 among seasons for the HF wetland treating wastewaters from a cheese plant in the north-west of Italy. Also, total nitrogen and organic nitrogen removals were quite consistent during the year. The only parameter affected by season was ammonia.

Wallace et al. (2001) pointed out that use of constructed wetlands in sub-freezing winter environments imposes a number of unique design and operation requirements. The authors summarized their observations from Minnesota and Iowa in two major points.

1. Properly-designed insulation of the wetland bed is effective in preventing freezing and resulting hydraulic failure. Relying on snow and ice cover does not provide reliable insulation during cold periods with limited snow pack.

2. The type of mulch insulation used can strongly affect the performance of the system. Only well decomposed organic material such as reed-sedge peat can be used without degrading treatment efficiency.

3. Introduction of a mulch layer will affect oxygen transfer rates and consequently also pollutant removal performance, and plant establishment. These factors must be addressed for successful application of HF constructed wetland technology in cold climates.

Chapter 6

TYPES OF WASTEWATER TREATED IN HF CONSTRUCTED WETLANDS

Constructed wetlands have long been used primarily for treatment of municipal or domestic wastewaters. However, at present, constructed wetlands are used for wide variety of pollution, including agricultural and industrial wastewaters, various runoff waters and landfill leachate (Kadlec and Knight, 1996; Vymazal et al., 1998a; Kadlec et al., 2000; Vymazal, 2006b; see also Table 1-1).

In the following section we try to summarize the major types of wastewaters which have been treated in HF constructed wetlands. Wastewaters and polluted waters are divided into five major groups: municipal/domestic, industrial and agricultural wastewaters, runoff waters and landfill leachate. Agroindustrial wastewaters such as those from wineries, distilleries, food and meat processing are included in industrial wastewaters despite these wastewaters are produced during processing agricultural products. Separate section deals with Endocrine Disrupting Chemicals which may occur in many various types of wastewater. The text does not deal with constructed wetlands for sludge dewatering.

6.1 Municipal and domestic wastewaters

HF constructed wetlands are commonly used to treat municipal and domestic wastewaters as both secondary and tertiary treatment stages (Table 6-1). In general, HF constructed wetlands are not used to treat raw municipal wastewater. In general, municipal and domestic wastewaters do not differ in quality, the difference is the source of wastewater: domestic wastewater denotes on-site (single house or cluster of houses) source while municipal wastewater denotes larger source – village or town. Kadlec and Knight (1996) listed the "typical" composition of municipal wastewaters: BOD_5 220 mg l^{-1}, COD 500 mg l^{-1}, TSS 220 mg l^{-1}, NH_4-N 25 mg l^{-1}, NO_x-N 0 mg l^{-1}, N_{org} 15 mg l^{-1}, TKN 40 mg l^{-1}, TP 8 mg l^{-1}. However, the concentrations vary widely as a consequence of water consumption which varies substantially in cities and villages and also among countries.

The results shown in Table 6-1 indicate that HF constructed wetlands have been successfully used to treat municipal wastewater with a wide range of inflow concentrations. Especially important is the fact that HF constructed wetlands can successfully treat wastewaters with very low concentrations of organics. It is well known that conventional treatment systems such as activated sludge cannot treat wastewater with such low organic concentrations. For details on treatment performance of HF constructed wetlands for municipal wastewater see Section 5.4.

6.1.1 Combined sewer overflows

Rousseau et al. (2005) pointed out that combined sewer overflows (CSOs) are becoming increasingly undesirable for river water quality considerations (Mullis et al., 1997) and multiple approaches have been adopted to reduce their impact (Zabel et al., 2001). Storm water detention tanks are a common preventive measure but at small-scale wastewater treatment plants they are unpopular with the water companies because they require additional site visits and attendance time. As a consequence, operating costs can increase considerably. Another drawback of detention tanks is the virtual absence of pollutant removal processes. This concept is therefore increasingly being abandoned in favor of storm water treatment facilities (Griffin and Pamplin, 1998). Whilst CSO treatment options are multiple (Geiger, 1998), constructed wetlands offer an ecological and cost-effective solution in rural areas to minimize CSO effects on the receiving water bodies (Scholes et al., 1999; Carleton et al., 2001). For example, in the United Kingdom, the use of HF constructed wetlands for CSO is quite common (Cooper et al., 1996) with the surface area of about 0.5 m^2 PE^{-1} (Green and Martin, 1996).

6.1.2 Linear alkylbenzene sulfonates

Linear alkylbenzene sulfonates (LAS) are the most widely used synthetic anionic surfactants. They account for 28% of the total production of synthetic surfactants in Western Europe, Japan, and the United States (Huang et al., 2004). Due to their frequent use in laundry and cleaning products, LAS are a common constituent of municipal and industrial wastewaters. As studies on LAS removal in constructed wetlands have been

Table 6-1. Treatment performance of HF constructed wetlands for treatment of municipal and domestic sewage. Chemical parameters in mg l^{-1}, fecal coliforms (FC) in log CFU 100 ml^{-1}. Values are mostly annual means.

Location	Country	Area	Flow	BOD_5		TSS		TP		TN		NH_4-N		FC		Ref.
		(m^2)	(m^3 d^{-1})	In	Out	In	Out	In	Out	In	Out	In	Out	In	Out	
Wigmore	UK	328	87	5.8	1.1	9.7	3.8					0.67	0.24			1
Onšov	Czech Rep.	2,100	92	5.9	2.7	12.0	5.2	1.3	1.0	17.9	10.7	5.2	4.2	5.4	4.4	2
Leek Wootton	UK	825	306	8.5	2.3	17.7	3.8					5.5	0.44			1
Bear Creek, AL	USA	2,035	14.9	9.4	1.0	72	3.5	6.6	0.45	52	9.9	10.5	2.7	5.3	1.0	3
Pisgah	Jamaica	90	0.9	27	13	57	13	9.6	0.4	40	1.6	5.8	0.4	5.6	2.2	4
Lifosa	Lithuania	3,780	180-400	51	7.8	30.6	12.2	11.2	9.6	9.4	7.4					5
Baggiolino	Italy	96	6	81	7.2	55	18	5.7	1.8	72	25			6.5	2.5	6
Uggerhalne	Denmark	2,640	103	115	6.0	158	6.4	4.8	4.8	22.5	16.8	17.3	12.5			7
Ondřejov	Czech Rep.	806	50	143	14.8	129	2.4	8.8	7.0	57	42.5	35.5	33	7.4	5.6	2
Holtby	UK	612	30	189	18.5	135	19					65.5	42.3			1
Koloděje	Czech Rep.	4,495	176	204	15	102	11	10.1	6.8			42.1	22.2	8.2	5.2	2
Hasselt-Kiewit	Belgium	896	23.3	232	6.0	196	9.0	12.4	4.0	81	29					8
Brondum	Denmark	437	8.1	330	16	392	10	21	14.3	74.5	43.3					9
Middleton	UK	168	10	390	25	116	21					63.2	15.7			1
Glavotok	Croatia	360	40	427	56	171	32	13.2	5.9	152	80			6.2	3.0	10
Carrión de los Céspedes	Spain	229	5.8	513	67	304	33	14.5	10.9	110	53	84	48.7			11
Agronomica	Brazil	450	6.6	979	19	224	104					49	16			12

1- CWA (2006), 2-unpubl. results, 3-Watson (1990), 4-Stewart (2005), 5-Gasiunas and Strusevičius (2003), 6-Pucci et al. (2004), 7-Kadlec et al. (2000), 8-VMM (2006), 9-Schierup et al. (1990a), 10-Shalabi (2004), 11-Sardón et al. (2006), 12-Philippi et al. (2006)

carried out in municipal wastewater, we included LAS removal description in this section.

Del Bubba et al. (2000) used a pilot-scale HF constructed wetland planted with *Phragmites australis* in Florence, Italy, to study LAS removal. At HLR of 3.7 cm d^{-1} the removal of LAS amounted to 94.2% and 99.7% for filtered and unfiltered samples with the respective outflow concentrations of 0.9 and 0.06 mg l^{-1}. When the inflow concentrations were increased the removal effect remained unchanged. The inflow/outflow concentrations were 65/1.8 mg l^{-1} and 278/0.18 mg l^{-1} for filtered and unfiltered samples, respectively. High LAS removal was observed even at temperatures as low as 5-9°C. Sulfophenylcarboxylic acids represent the primary biodegradation products of LAS and, among these, sulfonezoic acid is present at significant percentages.

Billore et al. (2002) found in a 300 m^2 pilot scale HF constructed wetland receiving the sewage from the Ravindranagar residential colony in Ujjain, India that the longer chain alkyl LAS homologues were degraded to a greater extent than that of the shorter alkyl chains. The removal of C_{10}, C_{11}, C_{12} and C_{13} LAS homologues were removed to 43.4%, 61.3%, 75.7% and 87.7%, respectively. Also Thomas et al. (2003) found in three HF constructed wetlands in the U.K. that the longer alkyl chain homologues were removed to a greater extent than the shorter alkyl chain homologues in the order $C_{13} > C_{12} > C_{11} > C_{10}$. This decrease has been found by other authors and has been attributed to the differences in the degree to which the homologues are adsorbed onto suspended particles and different biodegradation rates (Swisher, 1987; Painter and Zabel, 1989). The increased biodegradation rate and adsorption are related to the increased hydrophobicity due to longer alkyl chains (Swisher, 1987). Faster rate of degradation of the longer chain homologues has been confirmed for all LAS chain lengths from C_6 to C_{16} (Swisher, 1987; Terzic et al., 1992; Prats et al., 1993).

Huang et al. (2004) studied the removal of LAS in a pilot-scale HC CW in Barcelona, Spain. The authors concluded that the highest rates of LAS oxidation were observed in shallow beds where a more oxidized environment occurred. They also observed that biodegradation of LAS and sulfophenyl carboxylate biointermediates occurred under sulfate-reducing and mixed conditions, i.e. sulfate reducing and denitrification. C_{13} LAS homologues were generally removed to a higher extent than the shorter alkyl chain counterparts. These results supported those reported by Billore et al. (2002). The removal has also been found to be temperature and HLR dependent. The dependence of LAS removal on HLR and seasons was also reported by Thomas et al. (2003) and Kantawanichkul and Wara-Aswapati (2005).

6.1.3 Pharmaceuticals

The occurrence of pharmaceuticals and personal care products (PPCPs) in the environment is a newly recognized problem (Daughton and Ternes, 1999). PPCPs have been detected in wastewater treatment plants effluents at low parts-per-billion concentrations. Moreover, it is known that PPCPs are not completely removed in wastewater treatment plants, so variable concentrations can reach surface, ground and coastal waters (Kolpin et al., 2002). Matamoros et al. (2005) monitored the removal of PPCPs in HF constructed wetland at Las Franqueses, Spain (see Fig. 7-25). The compounds involved in the study were Ibuprofen, CA-Ibuprofen, OH-Ibuprofen, Naproxen, Diclofenac, Caffeine and Dihydrojasmonate. The comparison with the elimination of PPCPs through other types of wastewater treatment plants (activated sludge, biofilter) showed either similar or slightly better results for HF CW.

6.2 Industrial wastewaters

There are variety of industrial wastewaters which have been treated in HF constructed wetlands. The quality of industrial wastewaters varies widely with many wastewaters having very high concentrations of pollutants (Table 6-2). Therefore, treatment of industrial wastewaters nearly always requires one or more pretreatment stages.

Table 6-2. Examples of concentration of organics and suspended solids in various industrial wastewaters. Based on Chudoba et al. (1991), Kadlec and Knight (1996), Mancini et al., (1994), Billore et al. (2001), Herold et al. (2000), Perdomo et al. (2000), Junsan et al. (2000).

	BOD_5 (mg l^{-1})	**COD (mg l^{-1})**	**TSS (mg l^{-1})**
Coke plant effluent	1,000-5,300	2,000-10,000	
Refinery effluents	10-1,000	50-4,000	10-300
Pulp and paper production	100-1,200	500-1,000	500-1200
Tannery effluents	1,000-2,000	2,000-4,000	1,500-3,000
Pharmaceuticals	1,300-3,200	5,000-12,000	
Laundry	500	1,000	
Organic chemistry	400-20,000	800-50,000	100-300
Textile mills	75-6,300	220-31,300	25-24,500
Distilleries	6,000-65,000	4,000-120,000	1,000-17,000
Breweries	500-3,300	750-6,000	100-500
Soft drink	770	1,400	140-5,000
Vegetable and food processing	270-8,000	500-10,000	20-2,500
Meat processing	600	40-3,000	200-2,000
Starch processing	1,500-12,000	4,000-18,000	250-2,000
Yeast processing	7,000-21,000	10,000-30,000	50-2,400
Dairy/cheese factories	1,400-48,000	2,000-95,000	20-4,500
Olive oil mill	10,000-80,000	70,000-200,000	

6.2.1 Petrochemical industry

Wallace (2002a) used aerated HF constructed wetland to treat petroleum contact wastewater at Williams Pipeline Company terminal facility in Watertown, South Dakota, USA. The 1,486 m^2 wetland was designed to receive 1.5 m^3 d^{-1} on a seasonal (May-October) basis. To provide adequate oxygen transfer, a Forced Bed Aeration™ system was built (Wallace, 2001). The system was initially planted with *Phragmites australis* and overseeded with *Phalaris arundinacea*. Average influent BOD_5 and ammonia concentrations (at 20% of the bed length) were approximately 10,000 mg l^{-1} and 100 mg l^{-1} with respective effluent concentrations (at 80% of the bed length) averaging approximately 6 mg l^{-1} and 0.5 mg l^{-1}. Also BTEX was removed in the first 40% of the bed length due to enhanced volatilization as a result of the aeration system (Wallace, 2002a).

Ji et al. (2002) reported the use of HF constructed wetlands to treat heavily oil-contaminated water produced in Liaohe Oilfields, China (for details see China). Wood and Hensman (1989) reported the use of 2,000 m^2 HF CW filled with waste and coarse ash and planted with *Typha* at the inlet and *Phragmites* at the outlet for the treatment of petrochemical effluents.

Chapple et al. (2002) reported on the use of HF constructed wetlands for reducing the dissolved hydrocarbons in the runoff from a decommissioned oil refinery. Two out of four pilot wetlands (300 m^2 each) were filled with soil and two were filled with gravel. All beds were planted with *Phragmites australis*. The study focused on diesel range organics (typically reported as C_{10}-C_{40}). Both types of beds were successful in removing hydrocarbons but soil-based beds suffered substantially from surface flow. The authors pointed out, however, that because of the much higher cost of gravel it is likely that any future full scale system will contain a mixture of soil and gravel beds.

Yang and Hu (2005) used HF CW to polish oil-refinery effluents in Taiwan (for more see section 7.6.9). A 25 ha HF CW was built at Heglig, Sudan to treat about 60,000 m^3 d^{-1} of hydrocarbon contaminated water from the oil fields (Oceans-ESU, 2008; Fig. 6-1).

6.2.2 Chemical industry

One of the largest HF constructed wetlands in Europe was built in 1990 at the Air Products chemical works at Billingham, Teeside, United Kingdom (Sands et al., 2000). The plant is producing alcohols for the plastics and detergent industries, phenol/acetone and derivatives for plastics, detergents, pharmaceuticals and flame-retardant purposes and amines and derivatives for drugs, detergents, paper treatment, agrochemicals and animal feedstock additives. Seven beds planted with *Phragmites australis* with a total area of 49,000 m^2 is filled with soil.

Figure 6-1. HF constructed wetland at Heglig, Sudan planted with *Phragmites australis* treating hydrocarbon contaminated water from the oil fields. Photo provided by Oceans-ESU Limited, United Kingdom, with permission.

Dias et al. (2006) mentioned that in 1998, HF CW (1,500 m^2) was added to existing vertical flow constructed wetland to treat wastewaters rich in nitrates from the production of nitric acid in Estarreja, Portugal (Fig. 6-2). Industrial wastewater including those from chemical industry is treated in hybrid system including HF wetland in Yantian Industry Area in Baoan District, Shengzhen City (Wang et al., 1994, see also Fig. 4-53).

Figure 6-2. HF constructed wetland at Estarreja, Portugal treating wastewaters from nitric acid production. Photo by Jan Vymazal.

6.2.3 Pulp and paper industry

Pulp mill effluents are complex mixtures of wood-derived organics as well as some inorganic ions and compounds. In untreated effluents, the BOD_5 is high (generally in the range of 200 to 800 mg l^{-1}). Secondary treatment (by aerated lagoon or activated sludge) can be quite effective, reducing the BOD_5 to about 10 to 100 mg l^{-1}. The compounds responsible for the BOD_5 of untreated effluents are primarily simple sugars, organic acids (e.g., acetic) and alcohols (e.g., methanol). After treatment, the residual BOD_5 is largely caused by biological solids and some more refractory organic compounds (Thut, 1993).

Thut (1990b, 1993) used a 3,750 m^2 CW planted with to treat pulp mill effluent. The system was very effective in removing BOD with removal being consistently between 80 and 90%. Because the secondary-treated effluent delivered to the wetland was of a high quality with an average of about 10 mg l^{-1}, this resulted in the 1 to 2 mg l^{-1} range in the wetland effluent throughout much of the study period. Removal of TSS and ammonia was variable but in general, quite high. However, the wetland had no beneficial effect on color or adsorbable organic halide (AOX).

Hammer et al. (1993b) reported on the use of HF constructed wetlands for color removal from pulp mill wastewaters. The early color removal results were encouraging despite the concomitant export of BOD_5. The authors suggested that a treatment system for tannins and lignins should be designed to optimize environmental conditions and retention times to enhance fungal decomposition of complex organics, and incorporate similar components for further decomposition by bacterial populations. Since fungal populations require an attachment substrate, a vegetated sand or porous soil substrate is likely to simulate natural soil conditions and provide aerobic environment and hydraulic conductivity needed to enhance fungal growth.

Removal of phenol from pulp and paper mill wastewaters was studied by Abira et al. (2005) in Webuye, Kenya. The HF wetland with an area of 30.7 m^2 was filled with gravel to a depth of 0.3 m and planted with *Cyperus immensus, Cyperus papyrus, Phragmites mauritianus* and *Typha domingensis* (for more details see Kenya).

6.2.4 Tannery industry

Calheiros et al. (2007) evaluated the use of HF constructed wetlands for the treatment of tannery wastewaters in Portugal. Five experimental units were planted with *Canna indica*, *Typha latifolia*, *Phragmites australis*, *Iris pseudacorus* and *Stenotaphrum secundatum* and filled with Leca (Filtralite® MR 3-8). The units were loaded with two HLR, 3 and 6 cm d^{-1}. COD was reduced by 41-73% for an inlet organic loading varying between 332 and 1,602 kg ha^{-1} d^{-1} and BOD_5 was reduced by 41-58% for an inlet organic loading between 218 and 780 kg ha^{-1} d^{-1}. No significant differences in removal performance were observed among planted units. Also, an unplanted unit did not differ significantly from planted units. The use HF constructed wetlands for tannery wastewaters has also been evaluated by Küçük et al. (2003), Daniels (1998, 2001a,b) and Dotro et al. (2006).

6.2.5 Textile industry

Bulc et al. (2006) pointed out that the continuous dyeing of cotton and polyester fabrics or their blends of various ratios in the industrial production can be performed by pad-dry, pad-dry-bake, pad-steam and thermosol methods using reactive, vat and dispersed dyes (Shore, 1995). Colored wastewater, with high COD, TOC and pH values and low BOD_5 values, is aesthetically unpleasant. Several methods have been studied for purification of dye-house effluents, such as carbon adsorption, chemical precipitation and flocculation, oxidation, ozonation, ion extraction and membrane filtration, but each method has significant disadvantages: incomplete ion removal, high energy requirements, and production of toxic sludges (Golob and Ojstršek, 2005; Alkan et al., 2005).

Davies and Cottingham (1992) used HF constructed wetlands planted with *Phragmites australis* for the treatment of complex wastewaters from a group of textile processing and dyeing facilities with a dark blue-blue coloration caused by dye residues. The experiments were carried out in Melbourne, Australia, in 150 m^2 wetlands which had been used for domestic wastewaters for three years. The hydraulic loading rate of 9.6 cm d^{-1} was the same as for domestic wastewater. The visible colorization of the textile wastewater was reduced very quickly as it passed through the bed and disappeared after only 6 m of travel. Also, the suspended solids inflow concentration of 80 mg l^{-1} quickly decreased to less than 10 mg l^{-1} after about 15 m of travel and then remained more or less unchanged.

Bulc et al. (2006) reported the use of VF (40 m^2)-HF (40 m^2) constructed wetland to treat textile wastewaters in Slovenia. At the flow of 1 m^3 d^{-1} the average treatment efficiency amounted to 77% for COD, 57% for BOD_5, 5% for TN, 77% for organic N, 62% for sulfate, 87% for anionic tenzides and 85% for color. The system exported ammonia-N. Also, the alkaline pH of the wastewater was lowered to neutral. Constructed wetlands showed buffering capacity due to acids produced by microbial action; this was also reported by Mbuligwe (2005). Bulc et al. (2006) pointed out that constructed wetlands have a significant performance advantage with respect to COD, mostly due to filtering, sedimentation, and aerobic/anaerobic processes, although the reductive biodegradation of azo-dyes produces aromatic amines that can be biodegraded more easily under aerobic than anaerobic conditions (Mbuligwe, 2005). Baughman et al. (2003) reported 20-34% efficiency for the COD inflow of 50 mg l^{-1} and Winter and Kickuth (1989) reached 65-76% efficiency for the inflow of 1,400 mg l^{-1}. The BOD_5/COD ratios between 0.19 and 0.43 indicates biologically the hardly-degradable nature of textile wastewater and therefore, high BOD_5 removal cannot be expected.

6.2.6 Abattoir

Poggi-Varaldo et al. (2002) described 1,144 m^2 HF constructed wetland as a part of treatment system for wastewaters from an abattoir (slaughter house) in the State of Hidalgo, México. The system consisted of primary sedimentation, anaerobic lagoon and HF constructed wetland. In Table 6-3, treatment performance of the HF part is presented. The overall treatment efficiencies were 90%, 91%, 85% for COD, BOD_5 and TSS, respectively. Reduction of fecal and total coliforms amounted to 5.5 and 5.0 log units, respectively.

Table 6-3. Treatment performance of the HF constructed wetland as a part of the treatment system designed to treat abattoir wastewater in México. Data from Poggi-Varaldo et al. (2002).

	Concentration			**Loading ($g\ m^{-2}\ d^{-1}$)**		
	In ($mg\ l^{-1}$)	**Out ($mg\ l^{-1}$)**	**Removal (%)**	**In**	**Out**	**Removed**
COD	1,440	375	74	82	21	61
BOD_5	585	137	77	33	8	25
TSS	421	236	44	24	13	11
NH_4-N	112.3	101.8	9	6.4	5.8	0.6
NO_3-N	1.4	1.6		0.08	0.09	
Org.N	10.1	5.3	48	0.6	0.3	0.3
TN	124	109	12	7.1	6.2	0.9
PO_4^{3-}-P	9.6	9.2	4	0.55	0.52	0.03

Finlayson and Chick (1983) and Finlayson et al. (1990) reported on the use of HF CWs for the treatment of poultry abattoir wastewaters. The trench was 50 m long and 2 m wide; the front third was planted with *Eleocharis sphacelata,* the centre third with *Typha orientalis* and the last third was planted with *Scirpus validus* (= S. *tabernaemontani*). The removal of TKN and TP averaged over the period of operation 58% and 48%, respectively. The removal of both parameters was substantially higher as compared to control unplanted trench.

Lavigne and Jankiewicz (2000) reported on the use of 1,200 m^2 HF constructed wetland for slaughterhouse wastewaters treatment in Ecuador (for details see Ecuador).

6.2.7 Food processing industry

Vrhovšek et al. (1996) described the use of 156 m^2 HF CW to treat highly loaded wastewater from a food processing plant in Gradišče, Slovenia. The constructed wetland is composed of two beds planted with *Carex gracilis* and *Phragmites australis*. The system exhibited excellent removal of organics – average COD and BOD_5 inflows of 3,674 mg l^{-1} and 962 mg l^{-1}, were reduced by 92% and 89%, respectively. The concentration of orthophosphate in the inflow reached a maximum value of 4.6 mg l^{-1} and the removal efficiency varied between 92 and 99%. Ammonium inflow concentrations varied between 2.1 and 16.3 mg l^{-1} with an average treatment efficiency of 86%. Urbanc-Berčič et al. (1998) reported also other HF constructed wetlands used to treat food processing wastewaters (for details see section Slovenia).

Wallace (2002b) reported on the use of 189 m^2 HF constructed wetland with artificial aeration designed to treat cheese-processing wastewaters in Eichten Cheese, a small dairy in Minnesota. Results shown in Table 6-4 indicate a positive influence of aeration system on BOD and total-N removal. In this case, 305 meters of the aeration tubing was used.

Table 6-4. BOD_5 and total nitrogen removal in HF constructed wetland near Center City, Minnesota. *No aeration, **aeration system placed into intermittent operation, ***aeration system in full operation. Data from Wallace (2000b).

	BOD_5 (mg l^{-1})		**Total N (mg l^{-1})**	
	Inflow	**Outflow**	**Inflow**	**Outflow**
12/1997 – 4/1999*	2,076	1,369	91.2	51.0
4/1999 – 6/1999**	2,810	565	88.7	65.0
9/1999 – 3/2001***	3,140	99	186	49.0

HF constructed wetland was also used by Khalil et al. (2005) to treat cheese dairy farm effluent in southern France. The 200 m^2 bed was filled with the local brown loamy soil amended with calcareous gravel (4-8 mm), compost and iron oxides. The authors reported only on the early stages of operation so it is not possible to evaluate the treatment performance in a long-term run. The average treatment efficiencies over the first 8 months of operation amounted to 40%, 50%, 70% and 62% for TKN, BOD_5, TOC and TSS, respectively.

Gorra et al. (2007) reported on the use of HF constructed wetland for the treatment of wastewater from a medium size cheese-making plant in Aosta Valley, north-west Italy in mountain region at the altitude of 540 m. The wetland (Fig. 6-3) is a long (ca. 100 m) narrow ditch 1 m deep and about 2 m wide. The slope follows a natural terrain configuration. The wetland is divided into five sections filled with gravel, ground ceramic wastes, magnetite, zeolite and local soil supplemented with compost and marble sand. The average influent BOD_5, N_{org} and NH_4-N concentrations of 839, 176 and 22.7 mg l^{-1} were reduced to 130, 133 and 16.6 mg l^{-1}, respectively during the period summer 2003-spring 2005 (Gorra, pers. comm.)

Figure 6-3. HF constructed wetlands for treatment of wastewaters from a cheese-making factory in Aosta Valley, north-west Italy Photos by Roberta Gorra, with permission.

Mantovi et al. (2007) described the use of HF constructed wetlands to treat wastewaters from the production of Italian cheese "Parmigiano-Reggiano" (400 m^2, 10.5 m^3 d^{-1}) and "Grana Padano" (2,700 m^2, 70 m^3 d^{-1}). The treatment efficiency in both systems was very high and amounted to 94, 96, 98, 62 and 45% for TSS, COD, BOD_5, TKN and TP, respectively. Also, the reduction of vegetable fats and oils was very high – the inflow concentrations of 59 mg l^{-1} (Parmigiano) and 167 mg l^{-1} (Grana Padano) were reduced to 1 and 2 mg l^{-1}, respectively. Only for ammonia-N a slight increase in concentration was observed. However, the NH_4-N concentrations at the outflow were not exceedingly high, about 10 mg l^{-1}.

White (1994) reported on the use of a HF constructed wetland for seafood processor wastewater. The two wetlands were 1 meter wide, 4 meters long and filled with a 0.3 meter layer of crushed limestone (2.5 – 5 cm diameter). One wetland was planted with *Phragmites australis* and the other with *Spartina alterniflora*. Based on the experience from this system a two-stage HF system was built. The system was built with *Spartina alterniflora* (Saltmarsh cordgrass) and was fed a synthetic wastewater made of dry catfood dissolved in tap water, ammonium acetate and ammonium chloride. At HLRs varying from 1.28 to 4.27 cm d^{-1}, the inflow concentrations of BOD_5 and ammonia of 125 mg l^{-1} and 95 mg l^{-1}, respectively were reduced to respective outflow concentrations of 7-11 mg l^{-1} and 5-54 mg l^{-1}. The HLR had much greater influence on removal of ammonia than BOD_5.

A 70 m^2 HF constructed wetland (Fig. 6-4) was used to treat domestic (75%) and wastewaters produced by seasonal food processing (cheese, tomato sauce, apple and grape juice, olive oil etc., 25%) near Florence in

Tuscany, Italy Pucci et al. (2000). The wetland is filled with gravel (d_{10} = 8 mm) and planted with *Phragmites australis*. The treatment performance results are shown in Table 6-5.

Figure 6-4. HF constructed wetland at Poggio Antico, Tuscany, Italy for the treatment of wastewaters produced during the organic farming activities. Photo by Fabio Masi with permission from IDRA S.r.l.

Table 6-5. Treatment performance of HF constructed wetland Poggio Antico, Tuscany, Italy. Microbial parameters in CFU 100 ml^{-1}. Data from Pucci et al. (2000).

Parameter	**Concentration (mg l^{-1})**		**Loading (kg ha^{-1} d^{-1})**		
	Inflow	**Outflow**	**Inflow**	**Outflow**	**Removed**
COD	1,105	110	253	25	228
TSS	145	27.6	33.1	6.3	26.8
TP	17.1	14.1	4.0	3.2	0.8
NH_4-N	25.6	11.5	5.85	2.6	3.25
MBAS(Tensides)	6.1	0.96	1.4	0.2	1.2
Fecal coliforms	340,000	800			
Total coliforms	1,000,000	9,250			
Fecal streptococci	1,900,000	4,000			
Escherichia coli	200,000	550			

Gasiunas and Struselvičius (2003) and Gasiunas et al. (2005) presented the results from a 1,880 m^2 HF constructed wetland designed to treat meat-processing wastewaters in Lithuania (Table 6-6). The wetland is filled with sand and planted with *Phragmites australis*. The pretreatment unit consists of a 500 m^3 septic tank. de Zeeuw et al. (1990) used HF constructed wetlands for the treatment of wastewaters from potato starch processing.

Table 6-6. Treatment performance of the HF constructed wetland in Nematekas, Lithuania for meat-processing wastewater. Values in mg l^{-1}. Data from Gasiunas and StruseVičiene (2003) and Gasiunas et al. (2005).

	Raw wastewater	After pretreatment	Outflow
BOD_5	826	186	29.2
TSS	598	212	43
TP	41.1	15.8	8.8
TN	107	63.7	37.6

6.2.8 Distillery and winery

Billore et al. (2001) reported on the use of HF constructed wetland to treat the secondary treated distillery effluent from a private distillery, Associated Alcohols and Breweries, Ltd. at Khodigram village in the outskirts of Baraha town in Central India. The treatment system consisted of pretreatment chamber and four cell HF wetland with total area of 364 m^2 planted with *Typha latifolia* and *Phragmites karka* in cells 3 and 4, respectively. The authors pointed out that wastewaters from distilleries are extremely strong (see also Table 6-2) and conventional treatment systems are not able to achieve required discharge limits. The BOD_5 and COD concentrations in the distillery effluent even after the conventional secondary treatment amounted to 2,540 mg l^{-1} and 13,866 mg l^{-1}, respectively and, therefore additional treatment was necessary. The system achieved COD, BOD_5 TKN and TP reductions of 64%, 84%, 59% and 79%. The study indicated that constructed wetlands may be a suitable tertiary treatment option.

Winery wastewaters are characterized by the high content of organic (up to 45,000 mg l^{-1} BOD_5) and solids content, high acidity and large variations in a seasonal flow production (Shepherd et al., 2001; Masi et al., 2002, de Lucas et al., 2006). Also, the winery wastewaters are characterized by low N/C and P/C ratios. Detailed studies on the organic composition indicated that ethanol and sugars (fructose and glucose) represent more than 90% of the organic load. However, the wastewater also includes low amounts (about 0.1 – 5% of the total COD) of recalcitrant constituents (polyphenols and lignins), that could be difficult to degrade because of their structure as well as high molecular weights. These complex characteristics mean that different treatment alternatives had been proposed (e.g., Andreottola et al., 2002; Brucculeri et al., 2005; Moletta, 2005). de Lucas et al. (2006) mentioned that single biological treatments could present several problems due to the toxicity of the effluents mostly due to the presence of polyphenolic compounds (Petruccioli et al., 2002).

Constructed wetlands may offer an efficient low-cost, low-maintenance and energy alternative for wineries that have sufficient land available for a wetland creation. Constructed wetlands also have the advantage of being able to accept seasonal flow fluctuations without adversely affecting the functional aspects of the treatment system (Masi et al., 2002, Grismer et al., 2003).

Masi et al. (2002) described three constructed wetland systems in Tuscany, Italy designed to treat winery wastewaters. The system La Croche consists of Imhoff tank and a single HF bed (215 m^2) filled with gravel 5-10 mm (Fig. 6-5). The system Azienda Vitivinicola Ornellaia is a hybrid constructed wetland consisting of an Imhoff tank, two 90 m^2 VF beds, 102 m^2 HF bed (filled with 8-12 mm gravel), 148 m^2 FWS wetland and 338 m^2 pond. System Casa Vinicola Luigi Cecchi and Sons consists of 480 m^2 HF bed filled with 5-10 mm gravel and 850 m^2 FWS wetland (Fig. 6-5). All HF beds are planted with *Phragmites australis*. The treatment effect of all systems is excellent (Table 6-7).

Figure 6-5. HF constructed wetland Azienda Agricola La Croche (left) and HF-FWS hybrid constructed wetland Casa Vinicola Luigi Cecchi and Sons (right), both Castellina in Chianti-Siena, Italy, for treatment of winery wastewaters. Photos by Fabio Masi with permission from IRIDRA S.r.l.

Table 6-7. Treatment performance of constructed wetlands treating winery wastewaters. All values in mg l^{-1}. Data based on Masi et al. (2002).

	Cecchi			**Ornellaia**				**La Croche**	
	In	HFout	FWSout	In	VFout	HFout	FWSout	In	HFout
BOD_5	1,833	49	24.5	425	337	286	28.6	354	29.7
COD	3,906	131	84	1003	690	431	79	722	90
TSS	213	13.3	23.4	103	41.8	23.9	25.3		
TN	18.9	4.8	3.5					65.2	27.5
NH_4-N				26.6	8.7	4.7	2.7	46.7	21.4
TP	4.7	1.5	1.3	1.9			0.12		

Sheridan et al. (2006) described the HF constructed wetland for treatment of winery effluent in South Africa. The wetland has an surface area of 160 m^2, it is filled with dolomitic gravel and it is planted with *Phragmites australis*.

6.2.9 Lignite pyrolysis

Wiessner et al. (1999) reported the use of HF constructed wetland to treat lignite pyrolysis wastewater which had been stored for a long time in an effluent pond. Calculations showed that it should be feasible to use a HF constructed wetland system as part of a complex remediation program to improve overall water quality and in particular to remove large amounts of ammonia nitrogen.

6.2.10 Coke plant effluent

Jardinier et al. (2001) reported on the use of the pilot scale two-stage HF constructed wetland to treat coke plant effluents in France. The total area of the system was 24 m^2, the beds were filled with gravel and planted with *Phragmites australis*. The coke plant effluents have high concentrations of ammonia, iron and trace elements. The authors concluded that HF constructed wetland may be a valid method to substantially decrease nitrogen concentrations and also to retain some metals and PAHs.

6.2.11 Mining waters

Gerth et al. (2005) used a hybrid FWS-HF constructed wetland (total area of 1,400 m^2) to treat seepage water from uranium mining in Achlema-Alberoda, Germany. The system consists of four basins and a lagoon. Two of the basins were planted with *Carex* sp. and the others with *Phragmites australis*. The first and second basins were operated with surface flow while the following with a subsurface flow. The authors pointed out that different conditions are needed for removal of arsenic (aerobic) and uranium (anaerobic).

Pantano et al. (2000) reported the use of HF constructed wetlands to treat mining impacted groundwater with elevated metal concentrations in Butte, Montana, USA. The wetlands were effective in removing Cd, Zn and Cu while arsenic was released from the system and lead concentrations were not affected by the wetland (for details see section 2.7 on Trace elements).

There is huge number of information on heavy metal removal in FWS constructed wetlands (see section 4.1.4.5).

6.2.12 Laundry

Davison et al. (2005, 2006) reported the use of HF constructed wetland planted with a mixture of *Typha orientalis* and *Bolboschoenus fluviatilis* for the treatment of laundry wastewater (see Fig. 7-48).

6.3 Agricultural wastewaters

6.3.1 Pig farms effluents

Finlayson et al. (1987, 1990) reported on the use of HF Cw for the treatment of piggery wastewater. The treatment system consisted of two 36 m^2 HF 20 mm gravel-based beds planted with *Typha domingensis* with the addition of *Scirpus validus* at Cooper County Hog Farm, in Springs, Australia. The results for the period March 15 – August 20, 1982 are presented in Table 6-8. Numbers in Table 6-8 are interesting because the total amount of removed pollutants reflects the evapotranspiration which may lower substantially the volume of outflowing water. Therefore, the percentage removal was much higher for loads as compared to concentrations only.

Table 6-8. Treatment performance and comparison of the total amount of pollutants removed in a pilot HF constructed wetland treating piggery wastewater in Australia. Removal efficiency as percentage given in parentheses. Data from Finlayson et al. (1987).

	Concentration ($mg\ l^{-1}$)		**Total amount (kg)**	
	In	Out	In	Out
TSS	214	82 (62)	11.8	3.0 (75)
COD	642	378 (41)	35.3	13.8 (61)
TKN	257	190 (26)	14.3	6.95 (51)
TP	19.8	15.2 (23)	1.1	0.56 (49)

Wang et al. (1994) reported on the use HF constructed wetland for treatment of pig raising farm at Leping, south China. The system consists of screens, sedimentary pond, upflow anaerobic hydrolysis pond, HF constructed wetland and fish pond. The constructed wetland consists of four cells with a total area of 495 m^2 filled with a gravel 50-80 mm, 30-50 mm, 20-30 mm and 10-20 mm, successively. The average flow is 81 $m^3\ d^{-1}$. In Table 6-9, average treatment efficiency of the constructed wetland are given. The average inflow/outflow concentrations for the whole system were 15,000/99 $mg\ l^{-1}$, 8,000/58 $mg\ l^{-1}$ and 70,000/432 $mg\ l^{-1}$ for COD, BOD_5 and TSS, respectively.

Table 6-9. Average concentrations and loadings for HF constructed wetland, a part of a treatment system for pig raising farm wastewaters in south China. Data from Wang et al. (1994).

	Inflow ($mg\ l^{-1}$)	**Outflow ($mg\ l^{-1}$)**	**Efficiency (%)**	**Loading ($kg\ ha^{-1}\ d^{-1}$)**		
				Inflow	**Outflow**	**Removed**
BOD_5	1,073	142	86.8	1,756	232	1,524
COD	1,847	246	86.7	3,022	403	2,619
TSS	4,200	589	86.0	6,873	964	5,909

Gray et al. (1990) reported on the use of HF wetlands for the treatment of a combined stream of wastewater from a septic tank and farmyard runoff resulting from muck from pig housing being scraped across the yard. The HF constructed wetland with the surface area of 93 m^2 was filled with limestone chippings (3-10 mm) and planted with *Phragmites australis*.

Junsan et al. (2000) used a 4-stage HF constructed wetland with a total surface area of 449 m^2 for the treatment of pig farm effluents in China. The filters were filled with crushed stones 4-5 cm, 3-4 cm. 2-3 cm and 1-2 cm in stages 1, 2, 3 and 4, respectively and planted with *Monochoria vaginalis*. The system performed quite well at HLR of 18.3 cm d^{-1} (calculated for the whole system) – concentrations of BOD_5, COD and TSS were reduced from 1,038 to 124 mg l^{-1}, 1,865 to 246 mg l^{-1} and 558 to 51.5 mg l^{-1}, respectively.

Kantawanichkul et al. (2003) and Kantawanichkul and Somprasert (2004, 2005) used a combined VF-HF and HF-VF constructed wetlands to treat pig farm effluents in Thailand with TKN and COD concentrations of about 400 and 1,000 mg l^{-1}. The authors pointed out that both systems were quite efficient in removing TN, organic carbon, TSS and TP.

Strusevičius and Strusevičiene (2003) presented the results from a 50m^2 HF constructed wetland designed to treat pig-breeding farm wastewaters in Lithuania (Table 6-10). The wetland was filled with sand and planted with *Phragmites australis*. The pretreatment unit consisted of a 3-chamber septic tank.

Table 6-10. Treatment performance of the HF constructed wetland R. Visockas, Lithuania for pig-breeding farm wastewater. All results in mg l^{-1}. Data from Strusevičius and Strusevičiene (2003).

	Raw wastewater	**After pretreatment**	**Outflow**
BOD_5	578	315	17
COD	722	374	68
TSS	506	177	30
TP	14	9.6	0.8
TN	43	31	19.1
NH_4-N	20.5	16.6	11.6

Lee et al. (2004) used HF CW for the treatment of swine effluent at three hydraulic retention times (4.3, 8.5 and 14.7 d). Results showed in Table 6-11 indicate that the system responded well to the changes in HRT in removing suspended solids and organics. The average removal efficiencies in the three phases were: TSS 96-99%, BOD_5 86-91%, COD 77-84%, TP 47-59% and TN 10-24%.

Table 6-11. Inflow and outflow loadings (kg ha^{-1} d^{-1}) for a HF CW treating swine farm effluent in Taiwan. Calculated from Lee et al. (2004).

	HRT 4.3 d		**HRT 8.5 d**		**HRT 14.7 d**	
	Inflow	**Outflow**	**Inflow**	**Outflow**	**Inflow**	**Outflow**
TSS	624	7.2	312	12.6	303	3.5
BOD_5	474	67.3	247	23.4	119	9.7
COD	1,370	317	700	114	390	68.6
TN	262	237	125	93.5	69	55.4
TP	47	25.1	24	12.5	15	6.2

6.3.2 Fish farm effluents

Zachritz and Jacquez (1993) reported the use of HF constructed wetland for a treatment of recycled water from a geothermal aquaculture- high density finfish culture- in New Mexico. The system was filled with 5-8 cm rock and planted by *Scirpus californicus*.

Comeau et al. (2001) used two HF constructed wetlands to treat trout farm effluents at the Pisciculture du Lac William near St-Ferdinand, southeast of Québec city. Effluents from trout farms are typically 20 to 25 times more diluted than medium strength municipal wastewaters and even below municipal secondary treatment criteria with BOD_5, COD, ammonia and TP concentrations of about 15, 12, 1.3 and 0.3 mg l^{-1} (MENV, 1999). With respect to receiving water quality objectives, the most constraining element to remove from freshwater fish farm effluents is phosphorus (Comeau et al., 2001). Two beds (20 x 8.5 m each) were filled with crushed limestone (0 to 2.5 mm in one bed and 2.5 to 5 mm in the other) and planted with *Phragmites australis*. The authors concluded that the potential of HF constructed wetlands as an ecologically attractive and economical method for treating fish farm effluents to reduce solids and phosphorus discharge appears promising.

Naylor et al. (2003) reported on the use of experimental HF constructed wetlands to treat diluted sludge from a freshwater fish farm anaerobic digester. Pollutant removal was generally very good with planted wetlands (*Phragmites australis, Typha latifolia*) clearly outperforming unplanted units in term of BOD_5, COD, TKN, NH_4-N. The removal of TSS, NO_3-N PO_4-P and TP were comparable. Chazarenc et al. (2007) used a combination of a

HF constructed wetland (28 m^2) followed by static columns filled with electric arc furnace slag to treat effluent from anaerobic digester-sludge storage tank at the flow-through trout fish farm. The TSS, COD, TKN and TP concentrations of 120, 710, 29 and 25 mg l^{-1} in the storage tank effluent were reduced to respective values of 23, 43, 5.9 and 11 mg l^{-1} in the HF wetland effluent. Slag columns reduced the TP concentrations down to 2.3 mg l^{-1}.

Also Schulz et al. (2003) reported successful use of HF constructed wetlands to treat rainbow trout farm effluents in Germany under various hydraulic retention times (HRT, 1.5, 2,5 and 7,5 hours). TSS and COD were reduced by 95.8 – 97.3% and 64.1-73.8%, respectively, and demonstrated no influence of HRT. Total phosphorus and total nitrogen removal rates varied from 49.0% to 68.5% and 20.6% to 41.8%, respectively, and were negatively correlated with HRTs. Effluent purification was best ast HRT of 7.5 hours, but sufficient removal rates were achieved for shorter HRTs as well.

6.3.3 Dairy effluents

Mantovi et al. (2002, 2003) reported the use of a HF constructed wetland to treat dairy parlor effluent and domestic sewage in an isolated mountain rural settlement in the province of Reggio Emilia, Italy. The treatment system consisted of pretreatment in an Imhoff tank and two 72 m^2 HF cells filled with washed gravel 8-12 mm in one bed and with fine washed gravel 3-6 mm. In each wetland, the parts near the wastewater inflow and outflow points were filled with coarser gravel with a diameter of 8-35 mm. Both beds were planted with *Phragmites australis*. The results (Table 6-12) were quite promising and the authors pointed out that this technology is an appropriate treatment to reduce pollutants in wastewater from rural activities to values acceptable for discharge into surface waters.

Table 6-12. Treatment efficiency of the HF constructed wetland treating dairy parlor wastewater and domestic sewage (4.4 m^3 d^{-1} and 1.9 m^3 d^{-1}, respectively) at the Santa Lucia farm in Casina, province of Reggio Emilia, Italy. TC in log CFU 100 ml^{-1}. From Mantovi et al. (2003) with permission of Elsevier.

	Inflow (mg l^{-1})	**Outflow (mg l^{-1})**	**Removal (%)**
BOD_5	451 (241 – 887)	28 (2 – 98)	93.7
COD	1,219 (858 – 2,312)	98 (32 – 205)	91.9
TSS	690 (410 – 1,540)	60 (10 – 260)	90.8
TKN	64.7 (44.7 – 173.9)	33.3 (17.2 – 54.4)	48.5
NH_4-N	22.4 (10.3 – 36.4)	24.5 (11.2 – 35.3)	
Norg.	42.3 (24.1 – 138.2)	8.8 (0.5 – 25.1)	79.1
TP	12.8 (6.5 – 27.4)	5.0 (0.8 – 8.3)	60.6
TC	2.2 x 10^6	8.4 x 10^3	99.6

Gasiunas et al. (2005) presented results from a HF constructed wetland treating domestic wastewater and wastewaters from a dairy farm in Lithuania (Table 6-13). The surface area is 100 m^2, the bed is filled with semi-coarse sand and planted with *Phragmites australis*. Pretreatment consists of 3-chamber septic tank and the flow varies between 1.5 and 2.4 $m^3 d^{-1}$.

Table 6-13. Treatment performance of the HF constructed wetland A. Visockas, Lithuania for dairy farm wastewater. All values in mg l^{-1}. Data from Strusevičius and Strusevičiene (2003).

	Raw wastewater	**After pretreatment**	**Outflow**
BOD_5	920	453	28.7
COD	2,266	846	109
TSS	480	172	18.3
TP	30	21.5	12.6
TN	135	101	39.2
NH_4-N	96.5	70.6	28.7

Kern and Brettar (2002) reported on the use of the experimental HF wetland to treat dairy farm wastewater in Potsdam, Germany. The wetland had an area of 10 m^2 and was filled with gravel (2-8 mm), compost and sand (0.5-2 mm) in the upper layer. *Spartina pectinata*, *Phragmites australis* and *Carex acutiformis* were used as vegetation cover. The authors concluded that although environmental conditions for nitrifiers and denitrifiers were not suitable in winter, the results demonstrate a high efficiency of the wetland for treatment of dairy farm wastewater. Despite high inflow N concentrations (264 mg l^{-1}) a treatment efficiency of 85-90% could be maintained during the winter. The overall removal was 91.6% for NH_4 and 80.6 for N_{org}.

Hill et al. (2003) reported on the use of HF system to remove soluble P from an 800-head dairy farm in New York State. Eight 55 m^2 beds were filled with four different material: wollastonite tailings (by-product of a mining operations containing approximately 15% wollastonite and 70-80% garnet), Norlite (shale that has been crushed and fired; it is a construction material that is classified as a lightweight aggregate), limestone and soil. Over a 1.5 years, soil removed the most soluble P (53%), followed by Norlite (34%), wollastonite tailings (13%) and limestone (4%).

Drizo et al. (2006) used HF wetlands planted with *Schoenoplectus fluviatilis* to treat dairy wastewaters in Vermont, USA. The results indicated that constructed wetlands have a good potential for dairy farms wastewater management under cold climate conditions. The authors also tested the use of supplemental aeration (Wallace, 2001) and concluded that it had a considerable positive effect on the removal of all investigated water quality parameters (BOD_5, dissolved reactive P and NH_4-N).

The use of HF constructed wetlands to teat dairy farm wastewaters was also reported by Gray et al. (1990) in United Kingdom, Chen et al. (1995) in USA or Tanner (1992) in New Zealand.

6.4 Runoff waters

6.4.1 Highway runoff

Shutes et al. (2001, 2003), Pontier et al. (2003) and Revitt et al. (2004) described the use of HF CW for highway runoff treatment along the A34 Newbury Bypass in the United Kingdom. Copper, chromium nickel total aqueous metal concentrations, although low, were consistently removed with maximum efficiencies of 67, 70 and 87%, respectively, particularly in the summer. Zinc showed the highest aqueous metal concentrations and the generally positive removal by the constructed wetland system (maximum efficiency of 61%) correlates with the expected metal uptake by *Typha latifolia* nd *Phragmites australis*. The authors pointed out that the wetland had been designed as HF wetland, however, during the storms, the runoff water will over-top the substrate and therefore, it turns the system into surface flow part way through an intense storm event.

Bresciani et al. (2007) reported on the highway runoff treatment project for the highway connection Villesse-Gorizia in Italy. The project includes a total of 60 constructed wetlands along 17 km of highway. Each system consists of a first flush sedimentation tank, HF constructed wetland, wet pond and a final vegetated retention area. The projected total investment cost is about 6,000,000 EUR (25% for the remote control). The wetland cost is about 260 EUR per one meter of highway or 95,000 EUR ha^{-1} of drained area.

Shutes et al. (1999) developed design criteria for HF constructed wetlands for stormwater runoff. The wetland should ideally have a minimum retention time of 30 minutes for the design storm event. An ideal design should retain the average annual storm volume for a minimum of 3-5 hours and preferably for 10-15 hours to achieve a good removal efficiency. For the design storm, the following criteria were recommended:

-retention time:	24 h maximum
-aspect ratio:	1:1 to 1:2
-slope of base of wetland bed:	1% maximum
-minimum substrate bed depth:	0.6 m
-substrate:	0.15 m of soil over 0.45 m pea gravel
-hydraulic conductivity:	10^{-3} to 10^{-2} m s^{-1}

6.4.2 Airport runoff

Airport runoff contains de-icing and anti-icing compounds applied to the aircraft, runways and taxiways. The principal materials involved are ethylene, di-ethylene, and propylene glycols (Worrall et al., 2002). Although polymeric agents are added to retain the de-icing materials on aircraft, for example, it has been reported that some 80% of the applied liquid runs off within the airfield boundary (O'Connor and Douglas, 1993). The discharge of glycol-laden runoff to receiving waters can have significant impacts on their quality and ecological integrity by imposing high biochemical oxygen demands (Ellis et al., 1997). Worrall et al. (2002) calculated the amount of 51,000 kg of glycol as potential daily pollution from de-icing process at London Heathrow International Airport. After the trial reed beds experiment constructed in 1994 (Revitt et al., 2001) a full-scale system was completed in 2002 with the primary aim to treat de-icing compounds contaminated runoff from an extensive catchment of some 600 ha of runways, taxiways, cargo areas and terminal buildings. The system comprises a series of aerated balancing ponds combined with 2.08 ha of gravel-based HF constructed wetlands (Fig. 6-6) together with a kilometer of rafted reedbeds (see Fig. 4-33). The HF wetlands are divided into 12 hydraulically discrete cells with open cross-channels every 10 m length to reduce the incidence of short-circuiting within the cells. The whole system is planted with *Phragmites australis* grown from seeds collected in the locality of Heathrow Airport (Worrall et al., 2002). Results have been reported by e.g. Richter et al. (2004).

Figure 6-6. 2.08 ha HF constructed wetland as a part of the treatment system for runoff containing de-icing compounds at London Heathrow Airport. In front, there is an open water outflow zone. Photo by Jan Vymazal.

Karrh et al. (2002) reported on the use of HF constructed wetland for the treatment of anti/de-icing runoff built at Westover Air Reserve base in western Massachusetts. The construction of 0.24 ha system with a designed flow of 380 m^3 d^{-1} began in August 2001. In 1994, 5,500 m^2 HF constructed wetland of a Kickuth type was built to treat de-icing runoff water at Zürich-Kloten Airport (Röthlisberger, 1996). The runoff water with COD concentrations of 600-900 mg l^{-1} was treated to below 40 mg l^{-1} after two months of operation and below the discharge limit of 20 mg l^{-1} during several months of operation (Flughafen Direktion Zürich, 1999). De-icing runoff is also treated, for example, in HF constructed wetlands at Edmonton, Canada (Higgins and Maclean, 2002; Higgins and Dechaine, 2006) and Berlin-Schönefeld (Abydoz Environmental, 2005).

The catchment area at Edmonton International Airport is very large, and this, coupled with the airport's tight clay soil, result in very large amounts of stormwater runoff. The HF constructed wetland consists of 12 square gravel-filled cells with sides of 47.5 m each arranged in six trains of two cells each. Wetland surface area is 2.7 ha and design conditions for the wetland were for the treatment of stormwater runoff contaminated with up to 1,350 mg l^{-1} of ethylene glycol at flows of up to 1,500 m^3 d^{-1} (Higgins and Dechaine, 2006).

6.4.3 Greenhouse and nursery runoff

Prystay and Lo (1996, 1998) tested the potential use of a HF constructed wetland with a surface area of 254 m^2 for the treatment of low organic carbon, high nutrient wastewaters (TOC 21 mg l^{-1}, TP 126 mg l^{-1}, NH_4-N 38 mg l^{-1}, NO_x-N 240 mg l^{-1}) generated in the greenhouse operations in Canada. Preliminary results comparing the capabilities of FWS and HF wetlands to treat these wastewaters suggested that shallow FWS wetland (15 cm) provided superior treatment over deeper FWS wetland (30 cm) and HF wetland given a constant HRT. The authors suggested that the treatment efficiency appeared to be related to the organic carbon concentration in the system implying increased treatment efficiencies can be achieved as the wetland mature and larger litter layer accumulates.

Headley et al. (2001) noted that in New South Wales, Australia, the introduction of legislation to control runoff and charge for water used in agricultural production has encouraged commercial plant nurseries to collect and recycle their irrigation drainage. Runoff from a nursery typically contains around 6 mg l^{-1} TN (70% as nitrate), 0.5 mg l^{-1} TP (>50% as PO_4), and virtually no organic matter (BOD_5 < 5 mg l^{-1}, DOC < 20 mg l^{-1}). The authors tested HF pilot-scale units filled with 10 mm basaltic gravel and planted with *Phragmites australis*. TN and TP load removals were > 84% and > 65% respectively at HRTs between 2 and 5 days, with the majority of outflowing TN and TP being organic in form. For TN, a strong relationship existed between removal rate (g m^{-2} d^{-1}) and loading rate (r^2 = 0.995), while a weaker relationship existed for TP (r^2 = 0.47). It was estimated that a 1 ha nursery would require a reed bed are of 200 m^2 for 2 day HRT.

Merlin et al. (2002b) tested in Nimes, France HF constructed experimental units to treat tomato greenhouse drainage solutions with the mean nitrate-N concentration of 329 mg l^{-1}. Up to 70% of nitrate was reduced in *Phragmites*-planted units.

6.4.4 Agricultural runoff

Zhou et al. (2004) reported on the use of HF constructed wetland to treat agriculture stormwater runoff in China. Three 10 x 1.5 m beds were filled with gravel (30-50 mm, porosity 43.1%). Two beds were planted with *Phragmites australis* and *Zizania caduciflora*, one bed was left unplanted. The average TN inflow concentration was approximately 22 mg l^{-1} in which about 80% was nitrate, 10% ammonia and 10% organic nitrogen. Removal ot TN decreased with HRT and varied between 43 and 80% and 27 and 79% in *Phragmites* and *Zizania* beds, respectively. The unplanted bed exhibited only 33 to 65% removal.

6.4.5 Urban runoff

For treatment of urban stormwater runoff, FWS constructed wetlands are mostly used. However, there are some examples of the use of HF constructed wetlands as well. Geary et al. (2006) reported the use of HF constructed wetland to treat urban runoff at Blue Haven, Australia. The catchment area was 21 ha and the use of the catchment is 100% urban. The removal efficiency was very good – 66%, 95% and 72% for TP, TSS and TN, respectively. Constructed wetlands for urban runoff were intensively reviewed by Scholz (2006).

6.5 Landfill leachate

Infiltration of precipitation and migration of water through municipal solid waste landfills produce leachate that contains undesirable or toxic organic chemicals. The chemical quality of landfill leachate differs greatly from one landfill to another (Table 6-14) and fluctuates seasonally within an individual landfill. Leachate composition is waste- and site-specific depending on the waste type, landfill age, and amount of infiltrating water (Staubitz et al., 1989).

Leachate is generally colored (Fig. 6-7), anoxic and has high concentrations of total dissolved solids, COD, BOD_5, ammonia, phenols, benzene, toluene, chloride, iron, manganese, arsenic, heavy metals such as lead, cadmium, zinc or chromium but little or no phosphorus (Shuckrow et al., 1980, Staubitz et al., 1989, Kadlec, 1999; McBean and Rovers, 1999).

A series of phases is discernible in the decomposition of solid waste. Although the phases are variously defined (e.g., Rovers and Farquhur, 1973), there is general agreement on changes (McBean and Rovers, 1999). Phase I, or the hydrolysis and acidification phase, involving aerobic decomposition is typically brief, and lasts for less than a month. Once the available oxygen within the waste is utilized, except in the vicinity of the landfill surface, aerobic decomposition terminates (McBean and Rovers, 1999).

Table 6-14. Examples of landfill leachate chemical parameters.

	Esval[1] Norway	Bølstad[1] Norway	Fulton County[2] IN, USA	City Sand[2] MI, USA	Huneault[3] ONT, CAN	Monroe County[4] NY, USA
pH	7.1	7.5				6.9
Cond. (μS cm^{-1})	6,410	2,940			3,530	4,740
COD (mg l^{-1})	2,267	314	1,540	3,203	289	
BOD_5 (mg l^{-1})	735	63	390	312	39.9	70
TOC (mg l^{-1})	678	124				160
TSS (mg l^{-1})	147	180	7,840	241	51.2	
TP (mg l^{-1})	1.1	0.3	0.92			1.9
PO_4-P (mg l^{-1})	0.14	0.05				0.36
TN (mg l^{-1})	219	47	287			
NH_4-N (mg l^{-1})	0.2	3	284	2,074	12.8	230
NO_3-N (mg l^{-1})	179	40	3			
Cl^- (mg l^{-1})	964	80			366	490
HCO_3^- (mg l^{-1})	2,868	861				2,970
Na (mg l^{-1})	624	67			369	410
K (mg l^{-1})	241	50				269
Ca (mg l^{-1})	432	36				180
Mg (mg l^{-1})	90	17				160
Fe (mg l^{-1})	45	35	178		2.8	51
Mn (mg l^{-1})	4.4	0.7	2.7		1.67	
As (μg l^{-1})			25			
Cu (μg l^{-1})	20	8	269	23	430	30
Zn (μg l^{-1})	312	34	1,820	810	30	227
Pb (μg l^{-1})	8.1	7	220	13	30	13
Cd (μg l^{-1})	0.9	1	15	2		<1
Ni (μg l^{-1})	39	12	246	202		65
Cr (μg l^{-1})	39	5	164	216		13
Ba (μg l^{-1})	307	34				310
Hg (μg l^{-1})	0.2	0.3	<1			<0.2
Phenol (mg l^{-1})						56
Benzene (μg l^{-1})						5.5
Toluene (μg l^{-1})						22
Xylene (μg l^{-1})						45

[1]Mæhlum et al. (1999), [2]Kadlec (1999), [3]Sartaj et al. (1999), [4]Eckhardt et al. (1999)

Phase II begins with the initiation of activities of anaerobic and facultative organisms (involving acetogenic bacteria). They hydrolyze and ferment cellulose and other putrescible materials, producing simpler, soluble compounds such as volatile fatty acids and ammonia. This phase can lasts for years, or even decades and leachates produced during this phase are characterized by high BOD_5 concentrations (even over 10,000 mg l^{-1}), high BOD/COD ratios (commonly greater than 0.7), acidic pH (typically 5 to 6), high concentrations of ammonia (in the range of 500 to 1,000 mg l^{-1}), chloride, sulfate, calcium, magnesium, sodium and strong unpleasant odors (Kylefors et al., 1994; McBean et al., 1995; McBean and Rovers, 1999).

Figure 6-7. Landfill leachate is highly colored liquid, very often with unpleasant smell. Leiria, Portugal. Photo by Jan Vymazal.

Phase II also involves slower-growing methanogenic bacteria gradually becoming established and consuming simple organic compounds, with the production of a mixture of carbon dioxide, methane, and other trace gaseous constituents that constitute landfill gas. The transition from phase II to phase III can take many years, may not be completed for decades, and is sometimes never completed. In phase III, bacteria gradually become established that are able to remove the soluble organic compounds, mainly fatty acids, which are largely responsible for the characteristics of phase II leachates. Leachates generated during phase III are often referred as "stabilized", but in the life cycle of a landfill this is biologically the most active level. A dynamic equilibrium is eventually established between acetogenic and methanogenic bacteria, wastes continue to decompose and leachates are characterized by relatively low BOD values, low BOD/COD ratios, neutral pH levels, strongly reducing redox potential and low concentrations of volatile fatty acids. Phase IV and V occur when landfill becomes depleted of degradable organics (Mc Bean and Rovers, 1999).

HF constructed wetlands have been frequently used for landfill leachate treatment either as a single treatment unit or in combination with types of constructed wetlands (Table 6-15, Figs. 6-8 and 6-9). CWA database (CWA, 2006) includes 17 HF constructed wetlands for landfill leachate treatment in United Kingdom with wetland area up to 2,800 m^2. The use of HF constructed wetlands was also reported by Sanford (1999) from Ithaca, New York, USA or Sloop et al. (1996) from New Hanover County, NC, USA.

Table 6-15. Examples of HF constructed wetlands used to treat landfill leachate.

Country	**Location**	**Area (m^2)**	**Flow (m^3 d^{-1})**	**Reference**
Canada	Richmond, BC, Canada	6 x 45		Birkbeck et al. (1990), Experimental units
Norway	Esval	800	96	Mæhlum et al. (1999), part of a complex system
Norway	Bølstad	40	2	Mæhlum et al. (1999), part of a complex system
Poland	Szadółki	3,600	50	Obarska-Pempkowiak et al. (2005)
Slovenia	Ljubljana	270	10	Bulc (2006), part of VF-HF system
Slovenia	Dragonja	450	10	Bulc et al. (1996); Urbanc-Berčič (1997)
Slovenia	Mislinjska Dobrava	600	35	Urbanc-Berčič (1997)
Slovenia	Lubevč	275	11	Urbanc-Bečič et al. (1998)
United Kingdom	Monument Hill	1,800		Robinson et al. (1999)
USA	Tompkins Co., New York	720	8	Surface et al. (1993)
USA	Monroe Co., New York	209	0.6	Eckhardt et al. (1999), FWS-HF system
USA	Jones Co., Iowa	93	0.55	Nivala et al. (2005), Pilot-scale

Hunter et al. (1993) tested HF pilot-scale system to treat woodwaste leachates in British Columbia, Canada. Wood waste, referred to in the forest industry as hogfuel, is an abundant by-product and is commonly used as an inexpensive, light fill for road construction, horticulture, and animal husbandry. It is known to leach a highly colored, odiforous, and toxic liquid. Because this leachate has a high oxygen demand and can be toxic to fish, it is essential to treat it prior to discharging to fish-containing receiving waters (Hunter et al., 1993).

Figure 6-8. HF constructed wetland at Ljubevč, Slovenia for landfill leachate treatment. Two beds (150 and 125 m^2) are planted with *Phragmites australis*. Photo by Tjaša Bulc, with permission.

Figure 6-9. HF constructed wetland as a part of a VF-HF hybrid constructed wetlands for landfill leachate in Leiria, Portugal. Photo by Jan Vymazal.

6.6 Endocrine Disrupting Chemicals and special organics

Chemical substances that can interfere with the normal functioning of the endocrine system have been termed Endocrine Disrupting Chemicals (EDCs) (Keith, 1997). Masi et al. (2004) pointed out that the full list of EDCs includes a large range of anthropogenic organic compounds, such as phthalates, pesticides, polychlorinated biphenyls (PCBs), dioxins, polycyclic aromatic hydrocarbons (PAHs), alkylphenols, bisphenols and steroid estrogens (Birkett and Lester, 2003).

Several of these substances have been released in increasing amounts in the environment for decades, and due to their low degradation rate, a substantial increase of their background concentrations have been observed in various environmental compartments (Skakkebaeck et al., 2000; Tyler et al., 1998). In wastewater, EDCs have a variety of sources. A number of EDC classes (phthalates, pesticides, PCBs and bisphenols) are industrial products, worldwide used for several applications and are therefore common pollutants (Staples et al., 1997; Kupfer, 1975; Chen et al., 2002). Other EDCs compounds such as dioxines and PAHs are not commercial products, but are formed as by-products of various industrial and combustion processes; they are transported from atmosphere to soil and water bodies (Birkett and Lester, 2003). Alkylphenols are metabolites of their ethoxylate precursors, which are non-ionic surfactants used in many industrial, commercial and household functions (Del Bubba and Lepri, 2002). The presence of steroid estrogens in wastewater mainly arises from direct female excretion, in particular from pregnant females and women using oral contraception or hormone replacement therapies (Arcand-Hoy et al., 1998; Andrews, 1995).

Masi et al. (2004) monitored the removal of EDCs in a HF (160 m^2) – VF (180 m^2) constructed wetland treating wastewaters from a hotel in Florence, Italy. The hybrid system has been suitable for the removal of EDCs since 8 months from its construction, and the effluent could be reused according to the Italian regulations. Among EDCs, trace amounts of estrogens (17-α-estradiol and ethynyl-estradiol), PAHs (naphthalene, phenanthrene, fluoranthene and pyrene) and phthalates (diethyl, di-n-butyl and bis-2-ethylhexylphthalate) were found in inlet wastewater. All of these compounds were removed at high percentage (up to 100% for estrogens), with the only exception of bis-2-ethylhexylphthalate which was released by the HDPE liner.

Giraud et al. (2001) described the use of HF constructed wetland to treat water contaminated with polycyclic aromatic hydrocarbons (PAHs), particularly fluoranthene and the possible role of fungi present in these ecosystems. Out of 40 fungal species from 24 genera, fluoranthene was degraded efficiently by 33 species while only 2 species were able to remove anthracene over 70%.

Container nurseries apply pesticides and nutrients at various times throughout the year. Overhead irrigation systems are commonly used to water the plants daily. As much as 70 to 75% of this irrigation water runs off the packed gravel beds that the container plants rest on (Cabrera, 1997; Beeson and Know, 1991). This runoff may have significant concentrations of pesticides such as simazine, metolachlor (e.g. Mahnken et al., 1999). Removal of pesticides has been reported using FWS constructed wetlands (Table 4-14), but studies with the use of HF systems are limited. Stearman et al. (2003) used HF constructed wetland to treat runoff water from container nursery. Herbicides studied were simazine (Princep) [2-chloro-4,6-bis(ethylamino)-s-triazine] and metolachlor (Pennant) [2-chloro-*N*-(2-ethyl-6-methylphenyl)-*N*-2-methoxy-1-methylethyl-acetamide]. At a slower flow rate, which corresponded to lower mass loading and greater hydraulic retention time, a greater percentage of pesticides was removed. At 2.3 d HRT 62% of the applied herbicide was removed while at 5.1 d HRT 82% of the herbicide was removed. This is in agreement with literature data where the HRTs of 2 or 3 days were not long enough for some pesticide removal processes to significantly occur, such as microbial degradation (McCormick and Hiltbold, 1966; Erickson and Lee, 1989; DeLaune et al., 1997). Also planted cells were more efficient than unplanted cells. During the 2-year period, HF wetlands planted with *Scirpus validus* removed 82.4% metolachlor and 77.1% simazine compared with control cells without plants, which removed 63.2% metolachlor and 64.3% simazine.

Zachritz et al. (1996) reported excellent removal of benzoic acid (ca 90%) in pilot-scale constructed wetlands planted with *Scirpus validus* up to inflow concentration of benzoic acid of 80 mg l^{-1}. The vegetated units outperformed unplanted units.

Moore et al. (2000a) used the HF constructed wetland to treat condensate-contaminated groundwater at the Gulf Strachan Gas Plant, near Rocky Mountain House, approximately 200 km northwest of Calgary, Alberta, Canada. The groundwater contained between 15 to 20 mg l^{-1} of C_5-C_{12} hydrocarbons, including 50% BTEX compounds (benzene, toluene, ethylbenzene, xylenes). HF constructed wetland with a total area of 850 m^2 was planted with *Phragmites australis* and *Typha latifolia*. The wetland was supplemented with artificial aeration at the bottom of the bed to prevent freezing. With the use of aeration during the period November – May, hydrocarbons were completely removed. Without aeration (May-November), hydrocarbon removal efficiency in the wetland varied from 30% to 100%. Without aeration, temperature appeared to be a significant factor in the variable removal rates. The authors pointed out that the main removal mechanism appeared to be volatilization. Also Wallace (2002a) reported effective BTEX removal in HF constructed wetland in South Dakota (see Petrochemical wastewaters).

Behrends et al. (2000) described the use of HF constructed wetland to treat groundwater contaminated with explosives such as TNT (2,4,6-trinitrotoluene), RDX (hexahydro-1,3,5-trinitro-1,3,5-triazine), HMX (octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine), TNB (1,3,5-Trinitrobenzene), 2A-DNT (2-amino-dinitrotoluene) and 4A-DNT (4-amino-2,6-dinitrotoluene) at the Milan Army Ammunition Plant near Milan, Tennessee, USA.

Del Bubba et al. (1998) reported on the use of HF constructed wetland in tertiary treatment of municipal activated sludge system in Florence, Italy with respect to hydrocarbons removal. The wetland was quite efficient in removing aliphatic hydrocarbons (including n-alkanes), alkylbenzenes (C_2-C_3), naphatalene, nonylphenols and phthalates.

Braeckevelt et al. (2006) described the use of pilot-scale HF constructed wetland to treat contaminated groundwater with monochlorobenzene (MCB) in Bitterfeld, Saxony-Anhalt, Germany. At the site the groundwater is mainly contaminated by MCB with maximum concentrations of 22 mg l^{-1} (Vogt et al., 2002). The unit was filled with the local aquifer material consisted of sand with 30% gravel and 4% silt. The HLR was set at 1.9 cm d^{-1}. The wetland was planted with *Phragmites australis*. The results showed that MCB concentrations decreased along the wetland transect with the most effective removal in the upper layer. Isotopic fractionation provided evidence for *in situ* MCB degradation and suggested that anaerobic microbial degradation processes played a relevant role.

Chapter 7

THE USE OF HF CONSTRUCTED WETLANDS IN THE WORLD

7.1 Europe

7.1.1 Austria

Haberl et al. (1998) reviewed the situation concerning the use of constructed wetlands in Austria. They pointed out that due to lack of proper wastewater treatment plants in rural and mountain area CWs had been in discussion as an appropriate option for about 15 years in Austria. Several people promoted CW as on-site plants for these areas but at the same time regulatory authorities did not approve the CW technology. The major objections were the lack of long-term experience, winter operation, hygienic problems and clogging of the substrate. After emotional discussions several experimental CWs were built. One of the most extensively studied experimental plant of all times was CW at Mannersdorf built in 1982 (Fig. 7-1) - it had been monitored for 7 years (e.g., Haberl and Perfler, 1989, 1990). At the same year, HF system in Lainzer Tiergarten was put in operation.

The HF constructed wetlands turned out to be a very appropriate technology providing high stability in its efficiency with low levels of operation and maintenance. Removal of organics and suspended solids was efficient, however, HF CWs often proved insufficient as far as the removal of nutrients was concerned.

The survey carried out in early 1996 (Grabher, 1997) identified 160 operational constructed wetlands (59 VF, 45 HF, 8 VF-HF and 48 unknown systems) with Lower Austria and Styria being two federal counties in which

Figure 7-1. Experimental HF constructed wetland Mannersdorf, Austria. Photo by Raimund Haberl, with permission.

CWs were legally accepted. In other counties CWs were approved only as pilot plants. The authors pointed out that in these counties the approval by regulatory authorities could be expected within the next years. This apparently came true and at present, more than 1400 CWs are in operation in Austria (Mitterer-Reichmann, pers. comm.). The majority of constructed wetlands are designed with either vertical flow or as hybrid HF-VF systems (Mitterer-Reichmann, 2002).

Navara (1996) described the use of HF constructed wetlands for treatment of wastewater from a remote cabin at an altitude of 1 986 m in Austrian Alps (Table 7-1). The wetland is designed as a channel 300 m long with an average width of 0.7 m. The average slope is 10%. The system was planted with plants which fit into the alpine meadow environment, namely *Caltha palustris* (Marsh marigold), *Cardamine amara* (Bittercress), *Rumex alpinus* (Alpine dock), *Deschampsia cespitosa* (Tufted hairgrass) and *Carex rostrata* (Beaked sedge). The cabin operates for 110 days a year starting in mid June.

Table 7-1. Treatment performance of the system in the Ignaz Mattis Cabin in Austrian Alps at an altitude of 1986 m during the period 1989-1990. All values in mg l^{-1}. Data calculated from Navara (1996).

	BOD_5	**COD**	**TN**	**NH_4-N**	**TP**
Inflow	514	1,736	213	155	36.8
Outflow	6.1	42	27.4	2.7	0.61

Design parameters for HF constructed wetlands in Austria (ÖNORM B 2505, 2005):
Area: 4-6 m^2 PE^{-1}
Filtration material: inflow zone 16/32 mm, filtration bed 4/8 mm
Depth of the bed: 0.6 m
Hydraulic loading rate: 5 cm d^{-1}
Organic load: < 112 kg BOD_5 ha^{-1} d^{-1}

7.1.2 Belgium

One of the first reviews on constructed wetlands in Belgium appeared in 1998 (Cadelli et al., 1998). The authors pointed out that the use of constructed wetlands in the Walloon region was insignificant and only four CW_S were in operation during the survey and two of them were designed with a sub-surface flow. The system in Doische was built in 1989 and treats wastewater from 650 PE. The plant consisted of a stabilization pond and five beds in series (650 m^2 each). The vegetated beds consisted of a planted zone (89% of the area) followed by a zone of free water (11%) separated by a siphon separator (Godeaux, 1994; Vandevenne, 1995). The beds were planted with *Phragmites australis*, *Typha latifolia*, *Sparganium erectum*, *Iris pseudacorus*, *Carex acutiformis* and *Phalaris arundinacea*. The second CW with sub-surface flow was built in Lierneux in 1989 and served 500 PE. The systems consisted of four parallel *Phragmites* beds (60 m^2 each) and three *Scirpus* beds in series (270 m^2 each) (Godeaux, 1994).

Cadelli et al. (1998) in their summary pointed out that the development of extensive treatment technologies in Flanders was limited to a few installations incorporating a RBC followed by a reed bed. Overall, there was a lack of interest and great deal of wariness towards extensive technologies. However, the authors mentioned that the situation in Flanders was about to change rapidly as the company Aquafin (working with Severn Trent Water in UK), responsible for wastewater treatment planned to develop high-performance and economical alternative technologies for the treatment needs of a number of villages and residential areas which cannot be connected to existing or planned conventional treatment plants (Demil, 1996).

The survey carried out in Flanders by Rousseau et al. (2004b) identified 107 constructed wetlands of various types. However, only two constructed wetlands were those with horizontal sub-surface flow. The treatment system at Hasselt-Kiewit (Fig. 7-2) was put in operation in 1999, it is designed for 152 PE and the total area of 895 m^2 is divided into 8 parallel beds. HF system in Zemst-Kesterbeek (Fig. 7-2) treats municipal wastewater in two parallel beds with a total area of 1,300 m^2. This system was started in 2001. Treatment performance data are shown in Table 7-2.

Besides the two HF constructed wetlands described above, further ten HF systems were identified as a part of combined systems, mostly as a second part behind vertical flow beds (Rousseau et al., 2004b). Also, several HF constructed wetlands were identified as tertiary treatment units mostly behind rotating biological contactors.

Table 7-2. Treatment performance of two HF constructed wetlands in Flanders, Belgium. I = inflow, O = outflow, all data in mg l^{-1}. Data from VMM (2006).

	BOD_5		**COD**		**TSS**		**TN**		**TP**	
	I	O	I	O	I	O	I	O	I	O
Hasselt-Kiewit 2000	73	13	195	77	68	28	27.6	25.6	4.2	3.5
2001	126	6	338	57	301	12	58.3	14.6	9.7	2.8
2002	85	5	257	64	125	12	29.1	25.1	4.3	2.6
2003	162	4	418	52	157	11	59.1	24.7	9.8	3.5
2004	232	6	536	46	196	9	81.3	28.7	12.4	4.0
2005	110	4	383	42	41	8	38.1	23.1	6.3	3.7
Zemst-Kesterbeek 2001	18	2	103	28	75	7	12.2	7.6	2.1	0.9
2002	94	5	237	37	170	14	38.0	16.0	4.3	0.9
2003	105	4	275	32	71	11	54.8	23.8	5.7	1.7
2004	140	4	320	30	73	8	57.6	17.8	6.4	2.3
2005	87	3	380	31	41	8	64.4	18.2	8.3	2.2

Figure 7-2. Constructed wetlands Hasselt-Kiewit (left, photo by Diederik Rousseau, with permission) and Zemst-Kesterbeek (right, photo by Els Lesage, with permission) in Flanders, Belgium.

The Flemish guidelines (VMM, 2002) gives the following design parameters:
- at least 3 m^2 PE^{-1} for secondary treatment
- maximum length 15 m to prevent surfacing of water
- depth at least 0.4m at inlet (typical 0.6m), bottom slope 1%, max depth at outlet 0.8m
- influent distribution: equally over entire width, vertical risers to be preferred (adjustable over +/- 40 mm), distance between pipes 5-10 m.

- inlet: zone of about 0.5m filled with stones (60-100 mm diameter)
- filter material: hydraulic conductivity at least 3.6 m/h, washed gravel 5-10 mm recommended
- effluent: via drainage tube over entire width, situated in zone also filled with rocks 60-100 mm, drainage tube to be connected to elbow or flexible tube in order to be able to adjust water level.

Recently, HF constructed wetlands have been employed also in the Walloon part of Belgium. EPUVAL is the system conceived and developed by a non-profit association depending on the Agricultural University of Gembloux (Fonder and Xanthoulis, 2007). A standard design has been developed and applied - the treatment systems consists of a septic tank followed by two parallel HF wetland cells filled with gravel (10-20 mm and 3-8 mm) and planted with *Phragmites australis*. The system is designed with the Belgian standard of 120 liters PE^{-1} d^{-1} allowing 7.5 g BOD_5 m^{-2} d^{-1} of organic loading rate and HRT of 2.1 cm d^{-1}. Nominal hydraulic retention time is 4 days. The surface area for 5 PE is 28 m^2 (13.4 m (L), 2.1 m (W,)) for 10 PE the area is 56 m^2 for 20 PE it is 112 m^2, etc. The wetland cells are 0.8 m deep with water depth of 0.6 m. Eighteen of those systems have been built for private houses of 5 PE, 3 for 10 PE, 2 for 20 PE (Fig. 7-3), 2 for 40 PE (Fig. 7-3), 1 for 100 PE. Many studies have designed systems with various capacities, and are in progress. One specification is that owners can built the systems themselves under guidance of the engineers and technicians of the society (Fonder and Xanthoulis, 2007 and Fonder, pers. comm.).

Figure 7-3. HF constructed wetlands Nassogne for 20 PE (left) and Sainte Ode for 40 PE (right) in the Walloon region of Belgium. Photo by Nathalie Fonder, with permission.

7.1.3 Croatia

The treatment technology of constructed wetlands is relatively new in Croatia. Shalabi (2004 and pers. comm.) reported on several HF wetlands along the Adriatic Coast. System in Jakuševac (100 m^2) was built in 1999 and was designed to treat landfill leachate. A 360 m^2 HF system was built in

2000 to treat wastewaters from a camping Glavotok on the Island Krk. Both systems are planted with *Phragmites australis*. Other two systems are HF-VF hybrid systems: system in Zminj (built in 2002) has the area of 1700 m^2 and is planted with *P. australis* and *Juncus effusus*, the other system is an on-site system (built in 2004) with he area of 15 m^2 planted with *P. australis*.

7.1.4 Czech Republic

After a period of experiments in a small-scale constructed wetland (Vymazal, 1990), the first full-scale constructed wetland for the treatment of runoff waters from a dung-hill was built in 1989. Due to lack of rainwater and thus lack of runoff in summer 1989, it was decided to use the system for the treatment of sewage from the adjacent village. Despite the fact the treatment system was built with little knowledge of constructed wetlands and it was originally designed for different type of wastewater, the treatment effect was very high (Vymazal, 1998). However, the appearance of the system, which was really far from neat, became a pretence for negative opinions on constructed wetlands given by various organizations (hygiene service, water inspection, the Ministry of the Environment etc.). Unfortunately, the results themselves were not taken into consideration and, as a result, in 1990 none and in 1991 only four constructed wetlands for wastewater treatment were built and put in full operation.

Since 1992 a steep increase in a number of constructed wetlands has appeared (Fig. 7-4). The major factor influencing this phenomenon was the cancellation of “Recommended list of treatment systems for small point sources of pollution” at the end of 1991. This list offered various technologies (e.g. oxidation ditch, rotating biological contactors (RBCs) or simple activated sludge technologies) but did not include constructed wetlands (however, it did include soil filtration). Another reason for the fact that constructed wetlands were built in the Czech Republic after 1991 was the change in the socio-economic sphere. The towns and villages became much more independent - they handled their own budget and the decision-making rights became larger (Vymazal, 1998). By the end of 2006 about 180 constructed wetlands were put in operation (Fig. 7-4). All the constructed wetlands have been designed with horizontal subsurface flow. The constructed wetlands in the Czech Republic were continuously reviewed by Vymazal (1993, 1995b, 1996, 1998, 2001d, 2002a,b, 2006a).

Most constructed wetlands were designed to treat municipal or domestic sewage – 103 CWs for separate sewer effluents and 67 for combined sewer systems. Other uses include stormwater runoff, dairy, abattoir, bakery and goat farm (Vymazal, 2006a). The size varies from small systems for single houses to 1,400 PE. Most systems have been designed as on-site systems for less than 10 PE (50 systems, Fig. 7-5) and between 100 and 500 PE (77 systems, Fig. 7-6). Average specific area of vegetated beds is 5.7 m^2 PE^{-1} and average hydraulic loading rate is 5.2 cm d^{-1}. Filtration medium is washed gravel or crushed stones with fractions 4-8 and 8-16 mm. *Phragmites australis* is the most frequently used plant in the Czech Republic. It is used either as a single species or in combination with other species, such as *Phalaris arundinacea* (Fig. 7-7, Vymazal and Kröpfelová, 2005).

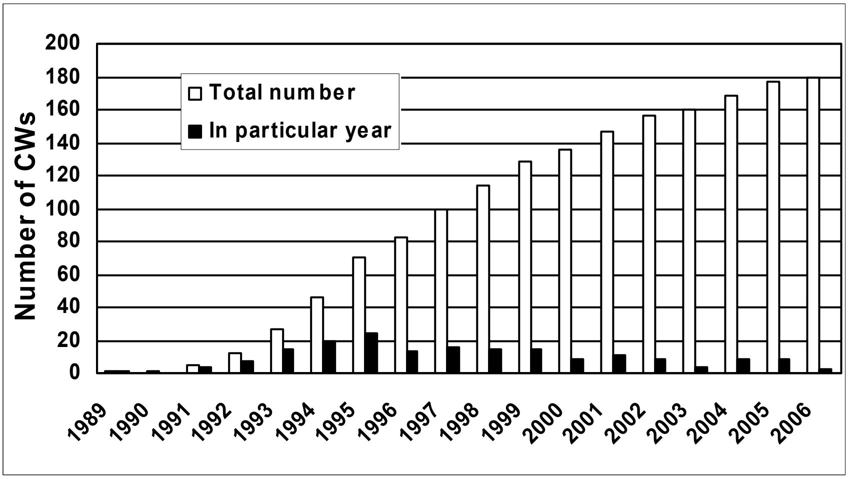


Figure 7-4. Constructed wetlands in the Czech Republic.

Figure 7-5. HF constructed wetlands for a single house wastewater treatment. Left: Kyjov (4 PE), right: Žitenice (6 PE). Photos by Jan Vymazal.

In the Czech Republic, wastewater treatment plants up to 2000 PE have discharge limits officially set only for BOD_5, COD and TSS. Therefore, most attention is paid to removal of these parameters. Table 7-3 indicates that HF systems in the Czech Republic achieve very good treatment effect for organics and suspended solids. Inflow organic loading is affected by the

Figure 7-6. Constructed wetland Břehov (100 PE) near České Budějovice, south Bohemia. Photo by Lenka Kröpfelová.

Figure 7-7. HF constructed wetland Čím planted with bands of *Phragmites australis* (left) and *Phalaris arundinacea* (right). Photo by Jan Vymazal.

fact that many constructed wetlands treat diluted wastewater from combined sewerage (Fig. 7-8). For example, Puigagut et al. (2007) reported for 13 Spanish HF wetlands an average inflow load of 105 kg BOD_5 ha^{-1} d^{-1}. However, the inflow value of 40.7 kg BOD_5 ha^{-1} d^{-1} is comparable with values reported from Denmark (48 kg BOD_5 ha^{-1} d^{-1}, Brix, 1994), Poland (37.7 kg BOD_5 ha^{-1} d^{-1}, Kowalik and Obarska-Pempkowiak, 1998) or Slovenia (53.9 kg BOD_5 ha^{-1} d^{-1}, Urbanc-Berčič et al., 1998).

Table 7-3. Removal efficiency of the HF constructed wetlands in the Czech Republic. *The numbers denote number of annual means taken into consideration with number of systems in parentheses. **First line: treatment effect of the whole system including pretreatment, second line: treatment effect of vegetated beds only. REM = removed load.

	Concentration ($mg\ l^{-1}$)		Eff. (%)	n*	Loading ($kg\ ha^{-1}\ d^{-1}$)			n*
	In	Out			In	Out	Rem	
BOD_5	162	14.7	84.1	321(61)				
**	131	17.0	80.1	124(35)	40.7	6.8	33.9	124(28)
COD	329	51	74.6	297(49)				
	238	59	67.4	118(33)	86.8	24.7	65.4	118(25)
TSS	159	12.7	80.6	314(62)				
	82	13	72.4	115(30)	30.7	4.4	26.6	88(26)
					Loading ($g\ m^{-2}\ yr^{-1}$)			
TP	6.9	4.0	33.1	185(47)				
	6.7	4.0	32.1	97(28)	106	68	38	75(22)
TN	52.1	25.6	44.5	66(23)				
	49.9	28.2	38.5	64(18)	803	490	313	56(16)
NH_4-N	30.5	18.1	28.4	219(56)				
	35.7	20.0	22.8	91(30)	496	310	186	70(25)

In the Czech Republic, HF constructed wetlands are frequently used to treat wastewater from combined sewer systems because they can successfully deal with low concentrations of organics and with fluctuating flow and inflow pollution concentrations (Fig. 7-8).

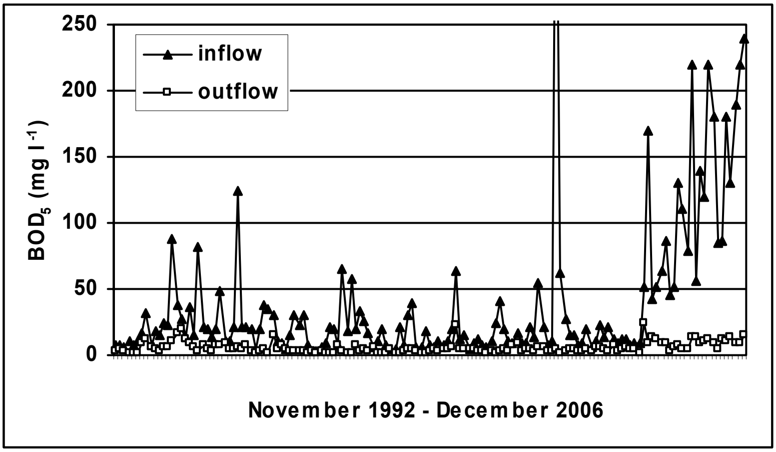


Figure 7-8. Removal of BOD_5 in HF constructed wetland Spálené Poříčí. Between 1992 and 2001 about 700 people were connected to a combined sewer system; in 2002, the constructed wetland was extended and additional 700 people were connected to separate sewer system.

7.1.5 Denmark

The initial Danish concept for construction of HF constructed wetlands had largely been a copy of the German ideas and recommendations, introduced in 1983 and the first full-scale treatment facilities in Denmark were constructed during winter 1983-1984 (see Fig. 5-2) (Brix and Schierup, 1989a,b; Schierup et al., 1990 a,b). This includes the inlet arrangement, area required, soil physical composition and the two preferred wetland plants *Phragmites australis* and *Typha latifolia* (Brix, 2003b). Several municipalities accepted the theoretical assumption for the operation of this kind of macrophyte-based wastewater treatment system concerning the reduction of BOD_5 and removal of total N and P, heavy metals and indicator bacteria. For all the parameters the efficiency >90% was claimed when a reed bed surface area of 3 to 5 m^2 PE^{-1} was assumed (Kickuth, 1980, 1982b). These claims were largely based on data from a system in Othfresen in Germany but the validity of these data proved to be doubtful (Brix, 1987a). There was a high interest in this technology shortly after the introduction of the technology to Denmark in 1983. The high interest prevailed until 1988 when the number of new constructions decreased (Fig. 7-9). This decrease was related to a new central legislation which defined new requirements to nutrient removal for large producers of wastewater (> 5,000 PE). In the following years until 1992, the municipalities therefore had to concentrate their investments on the major sources of wastewater, thus less emphasis was put on small systems (Brix, 2003b).

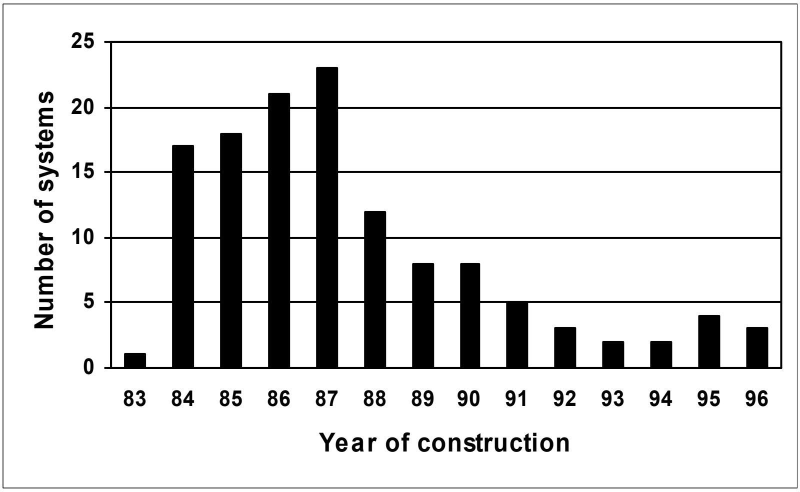


Figure 7-9. Number of HF constructed wetlands constructed in Denmark during the period 1983-1996. From Brix (1998) with permission from Backhuys Publishers.

The early Danish systems were very closely monitored (e.g., Brix and Schierup, 1989a,b; Schierup et al., 1990a,b; Willadsen et al., 1990). Inlet and outlet quality data from the Danish systems have been collected in a central database at the Department of Plant Ecology, University of Aarhus. The database contains results from a total of more than 7,000 samples from nearly 100 systems. Most systems were designed to treat wastewaters from small municipalities with few systems treating also dairy wastewaters (Fig. 7-10). The evaluation was done based on the total period of operation with the exclusion of the first two years of operation (Brix, 1998; Brix et al., 2006b).

Figure 7-10. HF constructed wetland Ingstrup, Denmark (100 m^2) built in 1984 to treat sewage and dairy farm wastewaters. Photo by Hans Brix, with permission.

The influent concentration of TSS varied between < 2 and 3,800 mg l^{-1} (median 75 mg l^{-1}) depending on the type of wastewater and the efficiency of the mechanical pretreatment. The concentration in the outflow were consistently low (median value of 7.6 mg l^{-1}) with 92% of all systems having median effluent concentrations below the general effluent standard of 20 mg l^{-1} and 76% systems having effluent concentrations below 10 mg l^{-1}. The median inlet BOD_5 concentration was 94 mg l^{-1} varying from < 10 to > 8,000 mg l^{-1}. About one third of all systems had median effluent concentrations < 5 mg l^{-1}, 80% had < 10 mg l^{-1}, and 98% had median effluent concentrations below 20 mg l^{-1}. The BOD_5 mass loading rate varied between < 1 and 23 g m^{-2} d^{-1} with an average value of 5 g m^{-2} d^{-1}. The effluent concentrations of TSS and BOD_5 were independent of the inflow concentrations (Fig. 7-11) as well as of the mass loading rate and hydraulic loading rate (Brix, 1998).

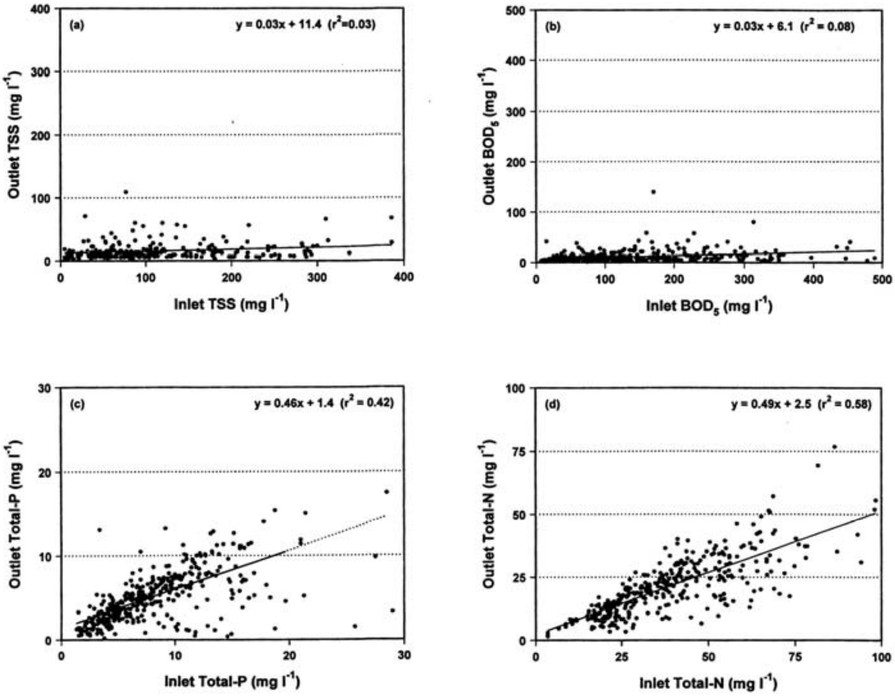


Figure 7-11. Annual mean inlet and outlet concentrations for 90 Danish HF constructed wetlands. TSS (n=257), BOD_5 (n=380), TP (n=362), TN (n=353). From Brix (1998) with permission from Backhuys Publishers.

The inlet concentrations of TP varied between 0.04 and 175 mg l^{-1} (median 7.4 mg l^{-1}), and the effluent concentrations varied between 0.01 and 55 mg l^{-1} (median 4.8 mg l^{-1}). The effluent concentrations of TP were linear related to the inlet concentrations (Fig. 7-11) but the scatter around the line is immense. The linear relationship between inflow and outflow concentrations was also observed for TN (Fig. 7-11). The median inflow and outflow TN concentrations were 35 (range of 2.1 to > 300 mg l^{-1}) and 20 mg l^{-1} (range of 0.7 to 198 mg l^{-1}), respectively. The median inflow and outflow NH_4-N concentrations were 25 and 12 mg l^{-1}, respectively.

Brix et al. (2006b) pointed out that effluent concentrations of TSS and BOD_5 generally decrease in the initial years as a result of development of dense vegetation in the beds. For nitrogen and phosphorus removal there seems to be little change in performance over time as the systems mature (Fig. 7-12).

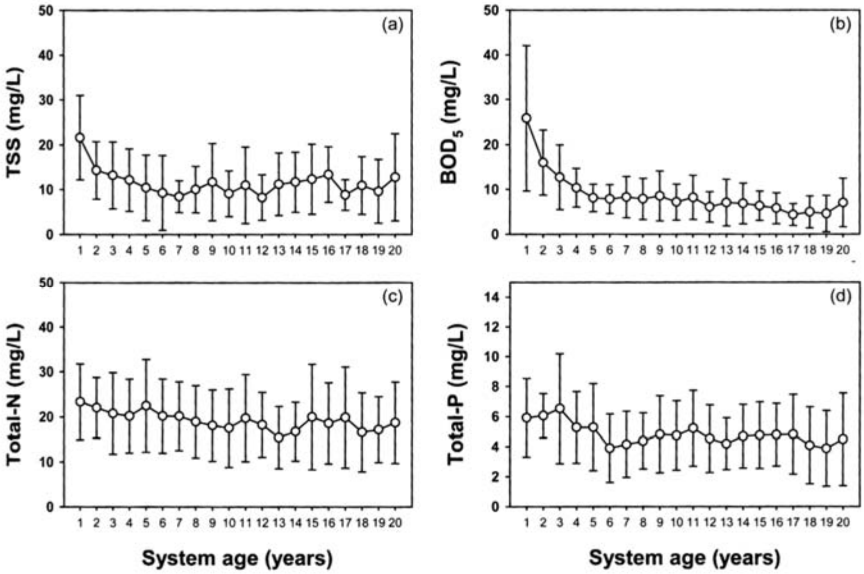


Figure 7-12. Annual average (± 1SD) outlet concentrations of a) TSS, b) BOD_5, c) TN and d) TP of 15 soil-based HF constructed wetlands plotted against system age. From Brix et al. (2006b) with permission of Portugal Ministry for Environment, Spatial Planning and Regional Development and IWA Specialist Group on the Use of Macrophytes in Water Pollution.

Following the adoption of Act 325 of 14 May 1997 on Wastewater Treatment in Rural Areas (Ministry of Environment and Energy, 1997) the Danish Ministry of Environment and Energy developed official guidelines for various treatment options for systems up to 30 person equivalents. The official guidelines include guidelines for soak-ways or soil infiltration, guidelines for root-zone systems which are equivalent to horizontal subsurface flow constructed wetlands (Ministry of Environment and Energy, 1999), and guidelines for biological sandfilters.

The major design parameters for HF constructed wetlands in Denmark could be summarized as follows:

- The sewage must be pre-treated in a two- or three-chamber sedimentation tank (minimum size 2 m^3 for a single household with up to 5 PE).
- The necessary surface area of the root-zone system is 5 m^2 per PE (minimum size for a single household is 25 m^2).
- The minimum length of the root-zone system is 10 m.
- The bottom of the bed should have a slope of 1% from inlet to outlet, but the surface of the bed should have no slope.
- Bed depth should be 0.6 m at the inlet side of the bed and deeper towards the outlet.
- Inlet and outlet zones consist of a transverse trench filled with stones securing that no wastewater are exposed to the atmosphere.

- The root-zone system must be enclosed by a tight membrane (minimum 0.5 mm thickness).
- The membrane must be protected by a geotextile or sand.
- The substrate must be uniform sand with a d_{10} between 0.3 and 2 mm, d_{60} between 0.5 and 8 mm, and the uniformity coefficient should be <4.
- The bed is planted with common reed (*Phragmites australis*)

The majority of root-zone systems in Denmark were built in the late 1980s and in the beginning of the1990s, and these were almost exclusively constructed with soil as the bed substrate and for the treatment of wastewater from small villages (Brix, 1998). The performance of these systems was good in terms of TSS and BOD removal, but no nitrification occurred and only a small fraction of phosphorus was eliminated. The new design where sand is used as the bed substrate (as described in the guidelines) is only used at very few sites because of the inability of the systems to nitrify and to remove phosphorus (Brix and Johansen, 1999; Brix et al., 2003).

7.1.6 Estonia

Mander et al. (2007) reported that in Estonia, there are 14 HF CWs and 10 hybrid systems containing HF CWs. Mander et al. (2001) and Kuusemets et al. (2002) extensively studied HF constructed wetland at Kodijärve treating wastewaters from a hospital (40 PE). The wetland consists of two beds (180 m^2 each) filled with coarse sand and planted with *Typha latifolia* in one bed and *Iris pseudacorus* + *Phragmites australis* in the other bed.

Öövel et al. (2007) reported on the HF constructed wetland as a part of a VF-HF hybrid system at Paistu treating wastewaters from a school. Area of the HF bed is 216 m^2 (Fig. 7-13), the bed is filled with 2-4 mm LWA (light weight aggregates) and planted with *P. australis*. HF wetland serves as a polishing unit and the final outflow concentrations are excellent (Table 7-4). Further information on HF constructed wetlands in Estonia could be found in Mander et al. (2003a,b, 2004, 2005) or Vohla et al. (2005a,b).

Table 7-4. Treatment performance of a HF constructed wetland as a part of VF-HF system treating wastewater from a school at Paistu, Estonia. Data from Öövel et al. (2007).

	Concentrations		**Loadings**		
	Inflow (mg l^{-1})	**Outflow (mg l^{-1})**	**Inflow (kg ha^{-1} d^{-1})**	**Outflow (kg ha^{-1} d^{-1})**	**Removed (kg ha^{-1} d^{-1})**
BOD_7	18.8	5.5	6.4	1.9	4.5
TSS	11.8	5.8	4.0	2.0	2.0
			(g m^{-2} yr^{-1})	**(g m^{-2} yr^{-1})**	**(g m^{-2} yr^{-1})**
NH_4^+-N	22.9	9.1	286	114	172
TN	36.1	19.2	451	240	211
PO_4^{3-}-P	0.46	0.16	5.8	2.0	3.8
TP	1.2	0.4	15	5.0	10

Figure 7-13. HF bed as a part of a hybrid system in Paistu, Estonia. Photo by Jan Vymazal.

7.1.7 France

Molle et al. (2005a) in their review on constructed wetlands in France indicated that HF constructed wetlands are mainly used as a part of various hybrid systems, however, they are used as a single treatment stage only rarely. In addition, most constructed wetlands in France are VF systems.

Merlin et al. (2002a) reported on the use HF CW situated in French pre-alpine mountain at the altitude of 720 m for 350 PE. The climate is characterized by an annual air temperature of 10°C varying during the year between -15°C to +35°C. The cold period represents about 5-6 months from November to April and during this period the mean effluent temperature was 8°C with temperature of 4.5°C being the lowest. The three stage system with a total area of 1,030 m^2 was planted with *Typha latifolia*, *Phragmites australis* and *Scirpus maritimus* in each stage. Evaluation after 6 years of operation revealed that removal of TSS amounted to 96% with more than 80% removed in the first stage (180 m^2). For COD and BOD_5, removal efficiency of the first stage approached 60% on average and reached nearly 94% in the outflow. The removal of TKN and TP amounted to 57 and 69%, respectively.

Merlin et al. (2002b) tested in Nimes, France HF constructed experimental units to treat tomato greenhouse drainage solutions with the mean nitrate-N concentration of 329 mg l^{-1}. Up to 70% of nitrate was reduced in *Phragmites*-planted units. The authors pointed out that constructed wetlands seemed adapted to treat drainage water of small tomato greenhouses. However, in order to obtain sufficient denitrification it is necessary to add an easily biodegradable carbon source and to maintain microaerophilic zones in the media.

HF constructed wetland was also used by Khalil et al. (2005) to treat cheese dairy farm effluent in southern France.

7.1.8 Germany

The history of the use HF constructed wetlands in Germany is very much the history of the use of this treatment system (see section 5.1). During the 1980s, HF constructed wetlands were intensively evaluated and discussed in the literature (e.g., Geller, 1984; Geller and Lenz, 1982; Bucksteeg, 1985, 1986; 1987a, b; 1990; Geller et al., 1990; Kickuth, 1984; Bucksteeg et al., 1985, 1987).

In 1989, national guidelines were published in Germany by ATV (Abwassertechnische Vereinigung) in 1989 (ATV H 262, 1989). The major points of these guidelines are summarized below (Bucksteeg, 1990). The guidelines define plant-based treatment systems as systems in which sewage is introduced into a soil matrix colonized by selected wetland plant species, over or through which the sewage flows in either a horizontal or vertical direction. The guidelines distinguished beds with a sandy or gravel matrix (suggested by Seidel, 1966; Seidel and Happel, 1983), beds with cohesive soil (suggested by Kickuth, 1981, 1984) or "flooded" beds (now called vertical flow wetlands). The constructed wetlands should receive only sewage which has been freed from coarse solids and sludge. For HF beds only cohesive soils with hydraulic permeability $\geq 10^{-4}$ m s^{-1} are able to provide horizontal passage of the wastewater below the bed surface. When using sand or gravel, coefficient of uniformity (d_{60}/d_{10}, see Section 5.2.3.2) should be ≤ 5. The effective particle size d_{10} must be ≥ 0.2 mm (Bucksteeg, 1990).

Börner et al. (1998) in their review evaluated the use of constructed wetlands in Germany with focus on Bavaria and Lower Saxony. The authors pointed out that most systems were HF wetlands with almost 3,000 systems in Lower Saxony. HF constructed wetlands are very often used for private households and other small sources of wastewater (Fig. 7-14). According to Wissing (pers. comm.), more than 50,000 small constructed wetlands were in operation by 2003 with majority of systems built to upgrade septic tank effluents.

7.1.9 Greece

Zdragas et al. (2002) reported on the use of HF constructed wetland as a part of FWS-HF hybrid system in Thessaloniki. Two beds (662 m^2 each) are planted with *Phragmites australis*. Four surface-flow beds (552 m^2 each) in the first stage are planted with *Typha latifolia* and the whole treats daily 100 m^3 of wastewater.

Figure 7-14. HF constructed wetland at Gasteig, Germany built in 1986. Photo by Zdeňka Žáková, with permission.

Tsihrintzis et al. (2004) described the combined VF-HF constructed wetland treating wastewaters in Gomati, Chalkidiki, northern Greece. The system consists of two stages of VF wetlands (640 and 360 m^2) and one HF bed (800 m^2) and planted with *Phragmites australis*. The hybrid system performed very well with BOD_5, COD, TSS and TKN removals of 93%, 78%, 95% and 68%, respectively. However, the system appeared to be a source of phosphate (-16.4%).

7.1.10 Ireland

Zhao et al. (2005) reported that according to a survey performed by University College Dublin, about 140 constructed wetlands sited were recorded in Ireland. Although the two typical types of constructed wetlands – FWS (see Fig. 4-21 and Table 4-11) and subsurface flow - are typically used, hybrid systems were also built. O'Hogain (2003, 2004) reported in detail on the use of 60 m^2 HF wetland as a part of VF-HF system at Colecott, Dublin County.

7.1.11 Italy

Until 1999, constructed wetland technology was not considered a treatment technology by the Italian legal framework and that was one of the main reasons why this technology has spread much less in Italy compared to other European countries (Masi et al., 2000). With the enforcement of a new law (D.Lgs. n.152 of 1999) which implemented EC Directive 91/271 about municipal wastewater treatment, constructed wetlands have been "officially" recognized as treatment technology. The use of constructed wetlands is especially advised for urban centers with populations in the range of 10-2,000 PE discharging into freshwater, in the range of 10-10,000 PE discharging in sea water, and tourist facilities and other point sources with high rates of fluctuation of organic and/or hydraulic loads (Masi et al., 2000).

Masi et al. (2000) and Puci et al. (2004) pointed out that constructed wetlands have been built in Italy since the mid-1980s but more wetlands were built only in the 1990s and 2000s, especially after 1999 when 23, 22 and 44 constructed wetlands were built in 1999, 2000 and 2001, respectively. Pucci et al. (2004) reported that about 175 constructed wetlands have been recognized in Italy with 75% of the systems being located in central and northern Italy (provinces Veneto, Emilia-Romagna and Toscana). HF systems are the most common (104 systems, 61%), various types of hybrid systems including HF wetlands represented 14% (23 systems). The size of the constructed wetlands in Italy is highly variable ranging from small on-site system to over 10 000 PE as tertiary treatment systems. Masi et al. (2000) noted that most HF systems were built either for secondary treatment of municipal/domestic wastewater (65%) and industrial wastewater (29%). Industrial wastewaters are mostly from food processing–(vegetables, oil, wine, cheese, beer), few systems were designed to treat car-wash wastewater or small breeding farms. According to Masi (pers. communication), the number of HF constructed wetlands in Italy increases exponentially mainly for HF systems < 50 PE. There are most probably over 1,000 systems in operation and about 300 systems in the public sector.

Pucci et al. (2004) summarized the major design parameters for HF constructed wetlands in Italy:

- area: 2-4 m^2 PE^{-1} for effluents discharged to surface waters
  4-6 m^2 PE^{-1} for effluents discharged to surface waters in sensitive areas, for reuse for irrigation or toilets
- filter material: gravel (4-16 mm), inflow zone: stones 80-120 mm

Conte et al. (2001) provided performance details on four HF constructed wetlands for municipal wastewater at Moscheta, Gorgona, Spannocchia and Pentolina (tertiary treatment) (Fig. 7-15, Table 7-5).

Figure 7-15. Four HF constructed wetlands in Tuscany, Italy. Top left: Gorgona Island, top right: Moscheta, Bottom left: Pentolina, bottom right: Spannocchia. For design details see Table 7-5. Photos by Fabio Masi, with permission of IRIDRA S.r.l.

Table 7-5. Performance of four HF constructed wetlands in Tuscany, Italy. MBAS = methylene blue active substances (tensides), TC = total coliforms, FC = fecal coliforms, FS =fecal stroptococci, EC = *Escherichia coli*. *IT=Imhoff tank, DG=degreaser, GC=grit chamber, AS=activated sludge. From Conte et al. (2001, reprinted from *Water Science and Technology* 44(11-12), pp. 339-343, with permission from the copyright holders, IWA.

	Moscheta		Gorgona		Spannocchia		Pentolina	
PE	150		350		60		500	
Area (m^2)	375		700		160		550	
Flow (m^3 d^{-1})	18		56		9		125	
Operation since	1999		1997		1997		1998	
Pretreatment*	IT+DG		GC+IT		IT+DG		AS	
	In	Out	In	Out	In	Out	In	Out
TSS (mg l^{-1})	308	5	139	73	31.5	18.2	114	14.5
COD (mg l^{-1})	543	30	252	95	175	46	600	78
NH_4^+-N (mg l^{-1})	41	5.9	26.3	15.3	22.8	9.0	58	31.7
NO_3-N (mg l^{-1})	0.84	0.6	4	0.8	16.1	9.3	1.5	3.7
PO_4-P (mg l^{-1})	11	6.1	11.7	5.6	2.44	1.97	3.4	2.2
MBAS (mg l^{-1})	15.8	2.0	3.2	1.2	0.14	0.08	15.4	2.5
TC ($\log_{10}$ CFU 100 ml^{-1})	8.2	4.6			4.5	3.2	6.3	5.5
FC ($\log_{10}$ CFU 100 ml^{-1})	8.2	4.5			3.4	2.8	5.1	4.9
FS ($\log_{10}$ CFU 100 ml^{-1})	4.6	3.0			3.4	2.1	5.4	4.8
EC ($\log_{10}$ CFU 100 ml^{-1})	7.8	4.3			2.0	1.0	4.9	3.5

In recent years, there have been many reports on various types of wastewater treated in HF constructed wetlands in Italy. Masi et al. (2002) described three constructed wetland systems in Tuscany, Italy designed to treat winery wastewaters (see section. 6.2.8). Mantovi et al. (2003) described the use of HF wetlands to treat dairy parlor wastewaters (see section 6.3.3). Mantovi et al. (2007) and Gorra (2007) evaluated the use of HF wetlands to treat wastewater from a cheese making factory (see also section 6.2.7). Del Bubba et al. (2000) evaluated the pilot scale experiments for LAS removal (see LAS). Albuzio et al. (2007) reported on the use of HF system for municipal sewage treatment in a mountain region (1000 m altitude) and Masi et al. (2006b) described wastewater treatment from tourist facilities in remote areas.

7.1.12 Lithuania

Gasiunas et al. (2005) evaluated the applications of HF constructed wetlands in Lithuania. Twenty HF constructed wetlands have been constructed since 1994. The HF wetlands are used for domestic wastewater as well as industrial wastewaters such as meat and vegetable processing industries and wastewaters from agroindustry (dairy farms, pig-breeding farms) (Gasiunas and Strusevičius, 2003; Strusevičius and Strusevičiene, 2003). The flow varied widely between 5 and 600 m^3 d^{-1}. For results see Tables 6-6, 6-10 and 6-13.

7.1.13 Netherlands

Veenstra (1998) pointed out that the introduction of constructed wetlands for wastewater treatment in the Netherlands materialized at the end of the 1960s, when the last of the three new polders in the former Zuiderzee was reclaimed. The Governmental Service Ijsel Lake Polder Authority/RIJP) decided to build several FWS constructed wetlands to treat domestic sewage from small isolated settlements and seasonally-occupied recreation resorts. Over the years the constructed wetland technology was also used to treat sewer overflows and wastewaters from small dairy farms. During the 1990s VF constructed wetlands were adopted for single house and small dairy form wastewaters (Veenstra, 1998, Van Dien, pers. comm.)

HF constructed wetlands are used in the Netherlands only seldom and are restricted to on-site treatment from individual households or small dairy farms. Also, a 638 m^2 HF system was designed to treat residential area stormwater runoff in Amsterdam (ECOFYT, 2007; Van Dien, pers, comm). de Zeeuw et al. (1990) used HF constructed wetlands planted with *Phragmites australis* for tertiary treatment of wastewaters from potato starch industry.

7.1.14 Norway

Jenssen et al. (2005) pointed out that the first constructed wetland with sub-surface flow for treatment of domestic wastewater was built in Norway in 1991. This method has become a popular method for wastewater treatment in rural Norway. Due to stringent discharge standards for phosphorus, the Norwegian concept for small constructed wetlands is based on the use of a septic tank followed by an aerobic vertical-downflow biofilter succeeded by a HF constructed wetland filled with media with high P-sorption capacity (Fig. 7-16). The aerobic biofilter, prior to the HF part, is essential to remove BOD and achieve nitrification. When designed according to present guidelines a consistent P-removal of >90% can be expected for 15 years using natural iron or calcite rich sand or a new manufactured lightweight aggregates (e.g., Filtralite P). When the media is saturated with P it can be used as soil conditioner and P-fertilizer. The results from 13 constructed wetlands are shown in Table 7-6.

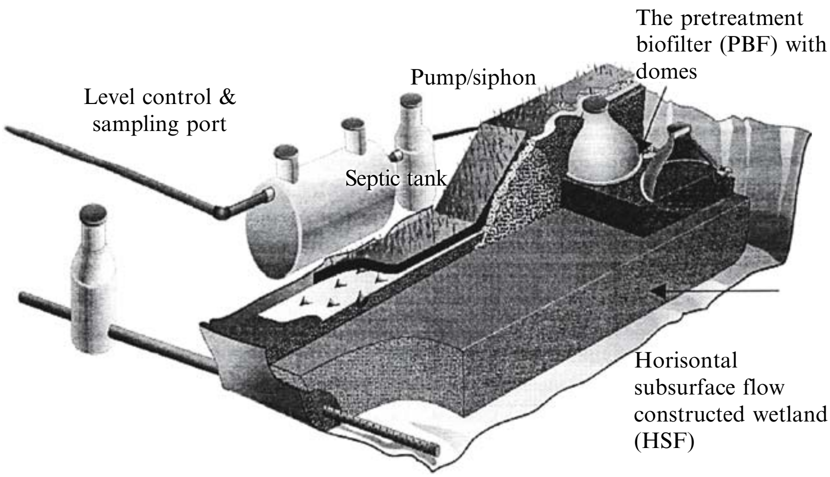


Figure 7-16. The last generation constructed wetlands for cold climate with integrated pre-treatment biofilter in Norway Jenssen and Mæhlum, 2003). With permission of Institute of Geography, Tartu University.

Table 7-6. Removal efficiencies and outflow concentrations in 13 constructed wetlands in Norway built between 1991 and 2000. Data from Jenssen et al. (2005).

	No. of systems	**Treatment Efficiency (%)**		**Outflow Concentration (mg l^{-1})**	
		range	mean	range	mean
BOD_7	5	80-98	89	5-22	13.6
COD	6	41-88	73	19-143	49
TN	9	41-79	61	2.5-49	19.4
TP	13	79-98	95	0.01-0.6	0.20

The systems built in Norway treating both domestic sewage (grey- and blackwaters) have a total surface area per person varying from 7 to 12 m^2 and according to present guidelines 7-9 m^2 is recommended. The depth of these systems is between 0.8 and 1.2 m and the guidelines recommend a minimum of 1 m. The reason is the cold climate and the need to meet the phosphorus discharge consent of 1 mg l^{-1} without frequent change of the P-saturated filter media (Jenssen et al. 2005). Design parameters with respect to cold climate were discussed by Jenssen et al. (1991, 1994).

Mæhlum (1994) described the use of HF constructed wetland as a part of a hybrid system for landfill leachate in Esval, Norway. The system consists of 400 m^3 anaerobic pond, 4,000 m^3 aerated lagoon with propeller aerators/mixers, two parallel 400 m^2 HF beds and 2,000 m^2 FWS constructed wetland. HF beds are filled with washed gravel and LECA (Light Expanded Clay Aggregates, 10-20 mm) and planted with *Phragmites australis* and *Typha* sp. The overall removal of organic matter, N, P, Fe and pathogens was promising (70 – 90%). Detailed performance results were presented by Mæhlum et al. (1999, 2002). Mæhlum et al. (2002) reported on another landfill leachate HF constructed wetlands at Bølstad.

7.1.15 Poland

Kowalik and Obarska-Pempkowiak (1998) reported that in the mid-1990s about 50 constructed wetlands were in operation in Poland. Most of the systems were HF wetlands varying from small on-site systems to large, secondary treatment systems up to 9,550 m^2.

Obarska-Pempkowiak et al. (2005) reported that at present, more than 100 constructed wetlands are in operation in Poland. Most of them are one-stage systems with horizontal subsurface flow, which are fed with sewage after mechanical pre-treatment. The authors pointed out that until 2002, all Polish standards were equal and obligatory for all wastewater treatment plants discharging more than 5 m^3 d^{-1}. The discharge limits for BOD_5, TN and NH_4-N were 30 mg l^{-1}, 30 mg l^{-1} and 6 mg l^{-1}, respectively (Regulations, 1991). The ammonia limit was difficult to meet for constructed wetlands. New Polish standards implemented in December 2002 (Regulations, 2002) were more liberal and more realistic. For wastewater treatment plants between 50 and 2,000 PE the discharge limits for BOD_5, COD and TSS are 40 mg l^{-1}, 150 mg l^{-1} and 50 mg l^{-1}. If the outflow is discharged to a lake, additional criteria for nutrients are set as follows: TN < 30 mg l^{-1}, TP < 5 mg l^{-1}.

Obarska-Pempkowiak and Sobocinski (2002) described eleven on-site HF constructed wetlands for 4-10 PE planted with *Salix viminalis* or *Phramites australis*. The surface area of the HF systems varied between 18 and 60. The length of all beds was 20 meters and width of the beds was variable, depending on the number of PE; 1.0 m, 1.1 m, 1.3 m and 1.5 m for

4, 5, 6 and 8 PE (Eymont, 1995). The beds were filled either with medium grain sand or a mixture of gravel (0.5 – 8 mm) and the artificial aggregate "Pollytag" (grain size 4-8 mm, porosity 40%) which is produced from the fly ash with an average composition: 58% SiO_2, 22% Al_2O_3, 1.4% Mg and 0.3% S.

The new discharge limits are usually met in a single stage HF constructed wetlands, however, for high removal of nitrogen hybrid constructed wetlands may be used. Obarska-Pempkowiak et al. (2005) and Obarska-Pempkowiak and Gajewska (2003) described the use of HF constructed wetlands within hybrid systems (Table 7-7, Fig. 7-17).

Information on constructed wetlands in Poland is very frequent in the literature (Bogdanowicz et al., 1996; Dzikiewicz, 1996; Ciupa, 1996; Kowalik and Obarska-Pempkowiak, 1998; Obarska-Pempkowiak, 1999, 2000, 2001, 2003; Obarska-Pempkowiak and Sobocinski, 2002; Obarska-Pemkowiak and Ozimek, 2003; Obarska-Pempkowiak and Gajewska, 2003, Obarska-Pempkowiak et al., 1994, 2005).

Table 7-7. Examples of HF constructed wetlands as a part of hybrid systems in Poland.

Locality	Type of wastewater		Area (m^2)	Reference
Wiklino	Sewage	HF-VF-HF	1,050/624/540	Obarska-Pempkowiak et al. (2005)
Wieszyno	Sewage	HF-VF-HF	600/300/600	Obarska-Pempkowiak et al. (2005)
Sarbsk	Sewage	HF-VF	1,610/520	Obarska-Pempkowiak et al. (2005)
Sobiechy	Sewage	HF-VF	448/88	Ciupa (1996)
Daržlubie	Sewage	HF-VF-HF	1200/500/1000	Obarska-Pempkowiak (1999)

Figure 7-17. Constructed wetland at Iwałdzie. Photo by Magdalena Gajewska, with permission.

7.1.16 Portugal

The history of the use of constructed wetlands in Portugal has been thoroughly reviewed by Dias et al. (2000) and Dias and Pacheco (2001). The authors pointed out that the interest in constructed wetlands started in Portugal around the late 1980s and early 1990s. At that time Käthe Seidel visited Portugal and tried to show and demonstrate the advantages of the use of CWs in the treating of wastewaters. Seidel's efforts did not generate immediate results, however. In the middle of the 1990s the installation of many constructed wetlands systems in Portugal began. This was due to two facts: a) the publication of a small brochure by Relvăo (1994) on adequate technologies for wastewater treatment of small communities, and b) the Reinhold Kickuth visit to Portugal and concomitant creation of a specific company to design and to help in the building of constructed wetlands systems. Besides that, between 1990 and 1993, various efforts were made to divulge CW technology in Portugal (dos Santos Oliveira, 1995; APDDA, 1998).

Small pilot-scale constructed wetland units have been in operation in the Lisbon State Institute, Evora State University and Fábrica Anilina in Estarreja since the late 1980s. The first full-scale constructed wetland (10,000 m^2, vertical flow) was built at Estarreja in 1994 (Novais and Martins-Dias. 2003) and was designed to treat industrial wastewaters (aniline and other chemicals). In 1998, HF constructed wetland (1,500 m^2) was added to treat wastewaters rich in nitrates from the production of nitric acid (Fig. 6-2). The first full-scale HF system for domestic wastewater was built in a prison facility at Pinheiro da Cruz near Setúbal (Fig.7-18). In 2000, the first HF constructed wetland was built to treat landfill leachate in Alcanadas (Fig. 7-19).

In their review, Dias and Pacheco (2001) listed 76 constructed wetlands built between 1993 and 2000 out of which 73 were HF systems and one VF-HF system. Most systems were designed for secondary treatment (3 - 1,160 PE, 10 - 2,900 m^2). Three tertiary treatment systems were designed for 2,500-5,000 PE with the area between 1,410 and 2,600 m^2. Most HF constructed wetlands were designed for 50-500 PE but on-site systems were also common (Fig. 7-20).

Figure 7-18. First full-scale HF constructed wetland for domestic wastewater in Portugal was built in Pinheiro da Cruz (200 PE, 578 m^2). Photo by Jan Vymazal.

Figure 7-19. Constructed wetlands (VF-HF-pond) in Alcanadas, Portugal designed for landfill leachate treatment. Photo by Jan Vymazal.

The Kickuth-type systems are filled with soil while other systems are filled either with crushed rock of gravel-sand. The major plants species used for HF constructed wetlands are *Phragmites australis*, *Typha latifolia* and *Iris pseudacorus*. Probably the largest system is in operation in Beja (see Figs. 5-39) where tertiary treatment HF system consists of 32 beds with a total area of 22,800 m^2. The average daily flow is 3.207 m^3 d^{-1}. (Dias et al., 2005).

In 2006, Dias et al. (2006) updated the situation and the survey identified 306 constructed wetlands with 176 being designed for individual households. Almost all constructed wetlands in Portugal have been designed with HF mode. Silva and Braga (2006a) reviewed constructed wetlands in the central Portugal. In 2004, 89 CWs served population of 42,500; in 2006

the number of CWs increased to 195 serving population of 80,100. The survey carried out in 2004 on 89 constructed wetlands revealed that:
- only for 4 systems were designed for industrial effluents while all others were designed to treat domestic wastewater;
- only 5% was used as tertiary treatment;
- 87 systems were HF wetlands;
- about half of the systems (45%) was designed for 200-500 persons and 11% of the systems were designed to treat wastewater from >1,000 persons;
- 78% of the systems have the area < 3 m^2 PE^{-1};
- the most frequently used plants were *Typha latifolia* (36%), *Phragmites australis* (29%) and *Juncus effusus* (20%).

Calheiros et al. (2007) evaluated the use of HF constructed wetlands for the treatment of tannery wastewaters in Portugal.

Figure 7-20. On-site HF constructed wetland at Malhão da Corcha near Odemira, Portugal planted with *Phragmites australis* (4 PE, 15 m^2). Photo by Jan Vymazal.

7.1.17 Slovakia

In Slovakia, HF constructed wetlands were built only as tertiary treatment systems in the 1990s. Between 1992 and 1998 seven HF constructed wetlands were built (Horvát, pers. comm.). The surface area of these systems varied between 440 and 900 m^2 serving population equivalent of 300-2,670. The first systems were designed to treat dairy wastewaters (2), municipal sewage (4) and one system was designed to treat washing waters from a train station. Halva and Števicová (2005) reviewed the use of HF constructed wetlands in Slovakia and reported 14 systems built between 1992 and 2005. However, only 9 systems were in operation at that time. The first system at Náplepkovo (Fig. 7-21) went out of operation because the dairy was closed. The only HF constructed wetland designed for secondary treatment of municipal wastewater was finished in 2006 in Zadný Hámor (Fig. 7-22). The system at Krásna Lúka (Fig. 7-23) has the largest area (1,500 m^2). All systems are planted with *Phragmites australis*.

Figure 7-21. HF constructed wetland at Nálepkovo, East Slovakia (500 m^2) designed for tertiary treatment of dairy wastewaters. The system built in 1992 is currently out of operation because the dairy was closed down. Photo by Jan Vymazal.

Figure 7-22. HF constructed wetland for secondary treatment of municipal sewage at Zadný Hámor, East Slovakia (17 PE, 90 m^2) after completion in November 2006. Photo by Jan Vymazal.

Figure 7-23. HF constructed wetland (700 PE, 1,500 m^2) for tertiary treatment of municipal sewage at Krásna Lúka, East Slovakia. Photo by Jan Vymazal.

7.1.18 Slovenia

Urbanc-Berčič et al. (1998) pointed out that the idea of wastewater treatment in the lagoons with floating plants and later in constructed wetlands was introduced in the late 1980s from Germany (Bucksteeg et al., 1987). The introduction of constructed wetlands was not systematic, since these systems were not considered to be acceptable technology. Most of the early systems were pilot systems for experimental work. Between 1989 and 1996, 17 systems were constructed throughout Slovenia with 14 systems being HF and three VF-HF wetlands (Urbanc-Berčič et al., 1998, Vrhovšek et al., 1996). The beds were mostly planted with *Phragmites australis* but other species such as *Juncus effusus*, *J. inflexus*, *Carex gracilis* and *Typha latifolia* were used as well. Then available results from HF constructed wetlands treating municipal sewage, wastewaters from food processing industry and landfill leachate are shown in Table 7-8.

Bulc et al. (2007) summarized that between 1989 and 2007, 28 constructed wetlands were constructed in Slovenia as HF or/and VF systems, operating in combination. 14 constructed wetlands were installed to treat sewage, 4 to treat industrial wastewaters, 7 for landfill leachate, 2 for conditioning drinking water, and 1 for motorway runoff treatment. The size of CW in Slovenia varies between 20 and 1,500 m^2, the depth varies between 0.4 and 0.9 m. For planting, rhizomes, seedlings or clumps of different wetland species were used but *Phragmites australis* the most frequently used plant species. Figure 7-24 features examples of Slovenian constructed wetlands.

There is a comprehensive information on the performance of HF constructed wetlands as single units or a part of hybrid systems treating landfill leachate (Urbanc-Berčič, 1992; Bulc et al., 1996; Bulc, 2006; Sajn-Slak et al., 2006; Zupanič Justin et al., 2007, see also Fig. 6-8).

Table 7-8. Average inflow (I) and outflow (O) concentration in HF constructed wetlands in Slovenia. M=municipal sewage, FP=food processing wastewater. LL=landfill lechate. Values in mg l^{-1}. From Urbanc-Berčič et al. (1998) with permission from Backhuys Publishers.

Location		**Period**	**Area**	**BOD_5**		**COD**		**NH_3-N**	
			(m^2)	**I**	**O**	**I**	**O**	**I**	**O**
Ajdovščina I	M	1990/93	250	76	8	165	34	5.5	2.0
Ponikva	M	1993/95	600	148	14	258	33	73.4	16.4
Kanal od Soči	FP	1991/95	56	126	62	500	267	9.6	5.5
Gradišče pri Kozini	FP	1991/95	156	823	85	3,381	455	7.8	1.0
Črnomejl	FP	1991/92	80	198	5	398	44		
Središče ob Dravi	FP	1991/93	80	320	61	1,143	276	4.3	3.1
Mislinjska Dobrava	LL	1990/93	600	60	23	268	163	291	236
Dragonja	LL	1992/95	450	100	51	1,297	589	489	156

7.1.19 Spain

Puigagut et al. (2007) in their review pointed out that in Spain, constructed wetlands were introduced in the mid-1990s but only during the 2000s the number of constructed wetlands substantially increased. In the survey carried out during the period January-June 2006, 39 subsurface flow constructed wetlands for the secondary treatment of municipal wastewaters were identified with most systems located in Catalonia (Fig. 7-25). It has been found that 87% out of 39 constructed wetlands are systems based on HF system alone or in combination with other unit processes. The specific area of HF constructed wetlands ranges between 1 and 7 m^2 PE^{-1} and most systems are planted with *Phragmites australis* with few systems planted with *Typha* sp. and *Salix* sp. The survey revealed that HF systems have been designed for 15 to 2,000 PE. In Table 7-9 the removal of organics and suspended solids is shown. In another survey in Catalonia, Velayos et al. (2006) identified 13 HF constructed wetlands mostly designed as secondary treatment of municipal sewage. Turon (2006) described in detail design parameters of 13 HF constructed wetlands and Vera et al. (2006) reviewed the situation in Canary Islands and Andalusia.

Figure 7-24. HF constructed wetlands for municipal sewage in Slovenia. Top: Ponikva, a 600 m^2 system built in 1992 (photo by Tjaša Griessler Bulc, with permission); middle: Velika Nedelja, a hybrid system consisting of HF bed (100 m^2), two VF beds (350 m^2 each), HF bed (100 m^2) and FWS pond (64 m^2) built in 2000 for 400 PE (photo by Tjaša Griessler Bulc, with permission); bottom: Sveti Tomaž, a 700 m^2 HF system with four beds for 250 PE built in 2001 (photo by Bogdan Macarol, with permission).

Table 7-9. Treatment performance of HF constructed wetlands in Spain. n = number of systems evaluated. Data from Puigagut et al. (2007).

	Concentration (mg l^{-1})			Loading (g m^{-2} d^{-1})		
	Inflow	**Outflow**	**n**	**Inflow**	**Outflow**	**n**
BOD_5	173	32	13	10.5	3.1	9
COD	347	84	13	22.6	8.6	9
TSS	173	22	3			

Despite constructed wetlands being the relatively new technology in Spain, many papers dealing with this technology have appeared recently. Gómez Cerezo et al. (2001) evaluated HF constructed wetlands as a part of multi-stage system for the tertiary treatment of lagoon system effluent at Mojacar, SE Spain. Huang et al. (2004) reported on the use HC CWs for removal of linear alkylbenzene sulfonates (LAS). Solano et al. (2004) reported on a pilot-scale system for a small village.

Figure 7-25. HF constructed wetland at Les Franqueses, Spain, treating wastewaters from 100 PE in 8 parallel beds (54 m^2 each). Photo by Joan García, with permission.

Garcia et al. (2004b) reported that the construction of a HF with a higher aspect ratio (length:width) and finer medium improves the hydraulic behavior of the system by reducing internal dispersion. Garcia et al. (2005) reported that water depth was a determining factor in the performance of the HF constructed experimental wetlands. Beds, with a water depth of 0.27 m in general removed more COD, BOD_5, ammonia and dissolved reactive phosphorus as compared to bed 0.5 m deep.

At present, there are no official guidelines for the design of HF constructed wetlands in Spain because most systems were built only recently (Puigagut et al., 2007). García et al. (2004b) and García and Corso (2007) suggested that the inflow BOD_5 loading should not exceed 6 g m^{-2} d^{-1}. Also, García and Corso (2007) proposed k_A 0.08 and 0.025 m d^{-1} for removal of BOD_5/TSS and ammonia, respectively, if the inflow BOD_5 concentration is < 250 mg l^{-1}. For higher BOD_5 concentrations, the k_A values should be lowered by 20%.

7.1.20 Sweden

Sundblad (1998) pointed out that as far as the constructed wetlands are concerned the major focus in Sweden is put on the use of surface flow wetlands with the aim to reduce phosphorus and nitrogen input into lakes and costal waters. More stringent demands for BOD and phosphorus removal in small communities with less than 100 inhabitants resulted also in the use of few HF constructed wetlands. Sundblad (1998) in her review reported at least six HF constructed wetlands and several more under construction. Figure 7-26 features HF constructed wetland treating wastewaters from a school.

In Table 7-10, examples of treatment efficiency of HF constructed wetlands in Sweden are shown. In general, the results are satisfactory for BOD_7 but not for phosphorus, for which the general Swedish requirement is 90% reduction or 0.5 mg l^{-1} in water discharged to the receiving water (Sundblad, 1998). Problems to achieve sufficient phosphorus removal, and the graduate decrease of BOD reduction in Höja has contributed to a lack of confidence in the HF constructed wetlands technology in Sweden. The Swedish E.P.A. suggested that an area of 20-25 m^2 PE^{-1} be used to assure sufficient removal of phosphorus (Lind, 1966). Sundblad (1998) pointed out that one conclusion from the Swedish experience would be that more development is obviously needed concerning the choice of substrate and application technique to increase phosphorus removal capacity (see section 5.4.4).

Table 7-10. Treatment performance of three HF constructed wetlands in South Sweden. All values in mg l^{-1}. From Sundblad (1998) with permission from Backhuys Publishers.

	BOD_7		TP		TN	
	IN	OUT	IN	OUT	IN	OUT
Snogeröd (1988-1991)	5.6	1.2	3.5	1.1	11	5.3
Fågeltofta (1991-1993)	136	4.6	4.7	1.4	26	15
Höja (1991-1994)	100	12	6.9	3.8	39	25

Figure 7-26. HF constructed wetland at Björkö-Ljungsbo for treatment of wastewater from a school with 250 children. Photo by Zdeňka Žáková, with permission.

7.1.21 Switzerland

Röthlisberger (1996) reported on three HF constructed wetlands based on the Kickuth design. These systems include municipal wastewater (115 PE), compost wastewater (180 PE) and constructed wetland for deicing runoff at Zürich-Kloten airport (5,500 m^2). The author also noted that there were six other HF systems (450 – 10,000 PE) either in construction in a design stage. However, no further information about these systems is available. Probably the best described wetland system in Switzerland was built in 1985 at the Centre for Applied Ecology in Schattweid (Schönborn and Züst, 1994; Schönborn et al., 1997; Billeter et al., 1998; Züst and Schönborn, 2003).

Züst and Schönborn (2003) reported that there are probably more than 150 constructed wetlands in Switzerland with no specification on the types. The authors also pointed out that constructed wetlands are of limited relevance in Switzerland where 92% of all inhabitants have been connected to sewer systems. However, there are still about 230,000 persons who have their wastewater treated by decentralized methods.

7.1.22 United Kingdom

The history of the use constructed wetlands in the United Kingdom was reviewed by Cooper et al. (1996), Cooper and Green (1998) or Cooper

(2003, 2006). The UK Water Industry first became aware of HF constructed wetlands (here called Reed Bed Treatment Systems) in mid 1985 when Water Research Centre (WRc) started to investigate the potential of the horizontal flow Root Zone Method (RZM) system which had then just started to be applied in Denmark following its development in Germany (Cooper and Green, 1998). The then Water Authorities were interested in a system which would allow them to apply low-cost, low-maintenance systems to small village communities which either had inadequate treatment or no treatment at all. Their interest was typically for its use in villages with populations from 50 to 1,000 PE (Cooper and Green, 1998).

On investigations it was found that the RZM process had been developed by Kickuth at the University of Hessen in Germany. The interest in the potential for the process led to a visit by a group of Water Authority and WRc personnel to talk to prof. Kickuth and see some of his systems in Germany (Boon, 1986, Fig. 7-27). Following the visit the WRc and Water Authorities staff were convinced that there was enough potential in the system to justify research and development work. It was however clear that there were several problems with the system that required solutions. To achieve rapid progress it was decided that all the authorities and WRc should work together and the Water Services Association Reed Bed Treatment System Co-ordinating Group was formed in late 1985 (Cooper and Green, 1998). The first systems went in operation in October 1985 (see Fig. 5-3) and by the end of 1987 about 25 full-scale systems were put in operation (Cooper and Boon, 1987; Cooper and Hobson, 1989). The major change in the design was the use of a flat surface and very coarse filtration material (5-10 mm fraction) which ensured sub-surface flow similarly to former design by Seidel (1965b). Also, specific area of 5 m^2 PE^{-1} was used in the United Kingdom.

The early systems were intensively monitored (e.g., Cooper and Boon, 1987; Cooper and Hobson, 1989; Coombes, 1990; Findlater et al., 1990; Upton and Griffin, 1990; Cooper, 1993; Cooper and Green, 1995; Cooper et al., 1990, 1997; Green and Upton, 1993, 1994, 1995; Green and Martin, 1996; Green et al., 1995). Gray et al. (1990) reported on the use of HF constructed wetlands for treatment of a pig farm runoff and dairy wastewaters.

Cooper and Green (1998) mentioned that the Severn Trent Water system at Little Stretton (Fig. 7-28) has been in operation since July 1987. It comprised 8 small beds - 12.5 m (L) x 2m (W) placed in series down a sloping site which allows gravity flux. The bed was one of the first constructed using gravel media and was the first to be planted with pot-grown seedlings. It was designed to treat up to 60 PE. The village had 40 residents and 20 PE was allowed for drainage known to come from the nearby dairy farm (at times during the first two years of operation the system had to treat up to 200 PE in BOD terms (Upton and Griffin, 1990). The performance of this system is documented in Table 7-11.

Figure 7-27. Visit of UK Water Authorities representatives to Beberbeck, Germany in July 1985. Photo by Paul Cooper, with permission.

Figure 7-28. HF constructed wetland at Little Stretton, Leicestershire, United Kingdom in July 1987 (left) and November 1987 (right). Photos by Paul Cooper, with permission.

Table 7-11. Annual average performance data from the Little Stretton HF constructed wetland between July 1987 and December 1995. From Cooper and Green (1998) with permission from Backhuys Publishers.

	BOD_5 (mg l^{-1})		**TSS (mg l^{-1})**		**NH_4-N (mg l^{-1})**		**$NO_{2,3}$-N (mg l^{-1})**	
	IN	**OUT**	**IN**	**OUT**	**IN**	**OUT**	**IN**	**OUT**
1987	147	29	132	19	10.0	10.0	15.0	1.0
1988	112	33	85	24	12.2	13.8	12.2	3.4
1989	162	34	127	43	14.9	11.3	9.1	3.0
1990	112	3.9	93	28	24.8	12.1	2.2	6.2
1991	55	4.1	70	28	14.7	5.9	9.0	6.2
1992	26	1.7	41	22	8.0	0.4	22.2	16.6
1993	35	1.7	30	8	8.4	0.2	16.5	11.4
1994	58	2.4	62	16	15.8	0.8	4.9	8.8
1995	78	7.3	65	16	19.5	3.6	6.1	5.1

Cooper and Green (1998) reported that by 1997 there were probably about 400 systems in the United Kingdom. In 2006, the Constructed Wetland Association (CWA, 2006; Cooper, 2006) recorded 956 constructed wetlands in the United Kingdom with 834 systems being HF systems. The majority of systems (677) served for tertiary treatment of municipal sewage (Fig. 7-29), 88 systems were designed for secondary treatment of sewage. In 2007, the number of constructed wetlands in the UK recorded in CWA database reached 1,012 (P. Cooper, pers. comm.) with estimated total number of 1,200 systems with the vast majority of the systems being built with horizontal subsurface flow. Shutes et al. (2005) in their review paper on urban surface drainage treatment systems in UK found that out of 103 systems 39 were constructed wetlands with 11 being designed with HF mode.

The Leek Wootton system (Fig. 7-30) was constructed in 1989/1990 to provide tertiary treatment to the effluent from a biological filter works covering a population of 1,150 from a village with two inns, a golf club and training college. It was designed at 0.7 m^2 PE^{-1} and the results are shown in Table 7-12.

The tertiary HF constructed wetlands are very common in the United Kingdom (CWA, 2006; Cooper, 2006). The tertiary beds at 1 m^2 per PE will achieve an effluent of < 5 mg l^{-1} BOD_5 and < 10 mg l^{-1} TSS and in many cases will achieve very substantial nitrification (Fig. 7-31). In Severn Trent where more than 350 tertiary beds are in operation, it has become standard practice to use 0.7 m^2 PE^{-1} for tertiary treatment (Green and Upton, 1995; Cooper and Green, 1998, Cooper, pers. comm.).

Figure 7-29. Examples of tertiary (a-c) and storm sewage overflow (d) HF constructed wetlands in the United Kingdom. Top left (a): Ashby Folville; Top right (b): Himley (Photos by Paul Cooper, with permission); Bottom left (c): Wormleighton; Bottom right (d): Leighthorne Heath – the front bed is flooded in order to protect newly planted *Phragmites* from rabbits; (Photos by Jan Vymazal).

Figure 7-30. HF constructed wetland for tertiary treatment of municipal wastewater at Leek Wootton. Photo by Paul Cooper, with permission.

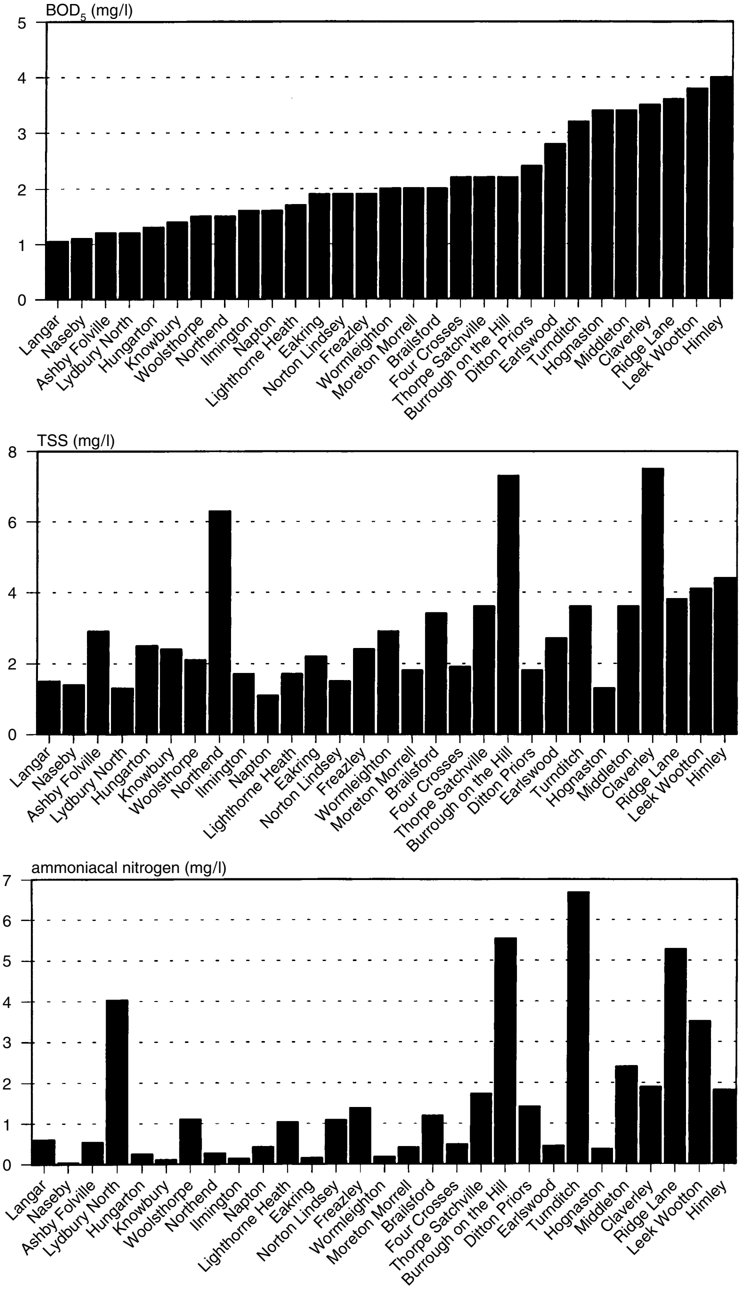


Figure 7-31. Average effluent quality from 29 tertiary HF constructed wetlands in Severn Trent Water in 1993 (January to December). From Cooper and Green (1998) with permission from Backhuys Publishers.

Table 7-12. Annual average performance (in mg l^{-1}) of Leek Wootton tertiary HF constructed wetland. From Cooper and Green (1998) with permission from Backhuys Publishers.

	BOD_5		COD		TSS		NH_4-N	
	IN	OUT	IN	OUT	IN	OUT	IN	OUT
1990/91	11.6	4.8	76	32	27.6	6.1	7.6	5.8
1991/92	11.9	2.0	77	34	19.1	3.7	5.4	1.9
1992/93	15.4	2.7	109	56	24.2	5.3	7.0	2.8
1993/94	9.1	1.5	94	49	16.3	4.4	7.2	3.0
1994/95	9.1	1.0	82	47	18.4	4.5	6.6	1.9

The treatment of storm sewage is one of the areas which has developed gradually since 1990 (Cooper and Green, 1998). The storm sewage overflow HF constructed wetlands (Fig. 7-29) are designed on the same basis as secondary and tertiary systems but with a lower area of 0.5 m^2 PE^{-1} and in cases where beds provide tertiary treatment as a continuous duty whilst taking storm sewage overflows as they occur an area of 1 m^2 PE^{-1} (Green et al., 1995; Green and Martin, 1996). Cooper (2006) pointed out that in UK database, 39 stormwater overflow treatment wetlands are included with 6 systems designed solely for storm sewage overflow and 33 systems designed in combination with tertiary treatment.

7.2 North America

7.2.1 Canada

Pries (1994) reviewed constructed wetlands in Canada but most systems were FWS constructed wetlands. Birkbeck et al. (1990) tested HF constructed wetlands planted with *Typha latifolia* and *Juncus effusus* to treat low-strength leachate in Richmond landfill, British Columbia. Six cells (45 m^2 each) were filled with dredged sand, sieved sand, 16 mm crushed gravel or 20 mm stone. The units were then covered with a topsoil. Because of operational problems only four cells were used for treatment evaluation. The test units removed between 59% and 78% of the BOD_5 and all four units removed more than 51% of COD producing an effluent with less that 20 mg l^{-1} of both BOD_5 and COD. Removal of ammonia was lower and varied among cells between 8 and 35%.

Moore et al. (2000) reported on the use of HF constructed wetland for treatment of groundwater contaminated with hydrocarbons including BTEX (for details see section 6.6).

HF constructed wetland (Fig. 7-32) was used as a first part of HF-FWS hybrid constructed wetland at “La Biosphère”, museum dedicated to to the study of all aspects of water, with a special focus on the Saint Lawrence River (Laouali et al. (1996). For details see Table 4-23 and Figure 4-50.

Figure 7-32. HF constructed wetland in front of the “La Biosphere” museum in Montréal. Photo by Jacques Brisson, with permission.

Comeau et al. (2001) reported on the use of HF constructed wetlands to treat trout farm effluents; Naylor et al. (2003) and Chazarenc et al. (2007) reported on the use of HF constructed wetlands to treat diluted sludge from a freshwater fish farm anaerobic sludge digester (see also section 6.3.2).

Prystay and Lo (1996, 1998) tested the potential use of a HF constructed wetland with a surface area of 254 m^2 for the treatment of low organic carbon, high nutrient wastewaters generated in the greenhouse operations. The authors suggested that the treatment efficiency appeared to be related to the organic carbon concentration in the system implying increased treatment efficiencies can be achieved as the wetland mature and larger litter layer accumulates (for more see section 6.4.3).

A 600 m^2 HF constructed wetland was built as tertiary treatment step for treatment of wastewaters from 137 mobile homes at Sunny Creek Estates in Bayside, Ontario (Anderson and Rosolen, 2000). The authors pointed out that wetlands are not normally approved as wastewater treatment facilities in Ontario and this wetland was operating as a demonstration site in an attempt to gain operational data needed for the assessment of constructed wetlands for wastewater treatment in cold climates.

A 2,600 m^2 HF constructed wetland was used as a part of the treatment system for landfill leachate in Moose Creek, Ontario, Canada (Kinsley et al., 2004, 2006). The whole system consisted of stabilization basin (10,000 m^3), peat filter (5,200 m^2), HF wetland with *Phragmites australis*, FWS wetland (3,600 m^2) planted with *Typha latifolia* and polishing pond (3,600 m^2).

7.2.2 Mexico

Belmont et al. (2004) described the use of HF constructed wetlands as a part of hybrid system in the community of Santa Maria Nativitas at a mean altitude of 2,380 m in central Mexico. The treatment systems consists of sedimentation terraces, stabilization pond, two parallel HF constructed wetlands (15.6 m^2 each, Fig. 7-33), VF constructed wetland (11.1 m^2) and two holding ponds (30 and 100 m^2). Both HF beds were filled with coarse gravel (3-5 cm), one cell was planted with Cattail (*Typha angustifolia*) and the second cell was planted with a mixture of Canna lily (*Canna* sp.) and Calla lily (*Calla* sp.). Treatment performance of the system is shown in Table 5-u. The treated water was suitable for irrigation, an important factor which could help to alleviate the scarity of water in central Mexico.

Table 7-13. Average concentrations for treatment system at Nativitas, Mexico during the period January to October 2001. Values in mg l^{-1}. Data from Belmont et al. (2004).

	Inflow	Terraces out	SP out	HF Cattail out	HF Mix out	VF out
COD	1,569	954	643	324	290	206
TSS	406	154	64.4	22.4	24.7	16.4
NH_4^+-N	66.3	44.4	43.8	42.4	39.3	21.6
NO_3^--N	28.4	14.3	11.2	6.6	6.5	5.5
TN	163	85.3	70	66.5	63.1	57.1

Figure 7-33. HF constructed wetland as part of the hybrid system at Santa Maria Nativitas in central Mexico. Photo by Marco Belmont, with permission.

Whitney et al. (2003) evaluated the performance of two-cell HF constructed wetland (total are of 81.2 m^2) serving 24 people in the coastal region of the Yucatan peninsula. The wetland was planted with as much as

58 plant species (Nelson, 1998) in 1996 but by 2001 less than 20 species were present. Unfortunately, the authors did not provide any results.

Poggi-Varaldo et al. (2002) described 1,144 m^2 HF constructed wetland as a part of treatment system for wastewaters from a abattoir (slaughter house) in the State of Hidalgo, México (for details see section 6.2.6). Additional data on the performance of this system were reported by Rivera et al. (1996).

A 75 m^2 HF constructed wetland (Fig. 7-34) was put in operation 1998 to treat highly contaminated surface water is treated with the goal to use it for irrigation of a tree nursery (Haberl, 1999; Durán-de-Bazúa et al., 2000).

Figure 7-34. HF constructed wetland in the Mexico City park "Viveros de Coyoacán" for the treatment of highly polluted Magdalena River. Photo by Carmen Durán-de-Bazúa, with permission.

7.2.3 United States

The first experimental small size HF mesocosms were built in 1972 in Seymour, Wisconsin and the pilot-scale HF constructed wetlands were built there in 1974. The beds of a pilot system were 19.3 m long, 3.05 m wide at the bottom and 5.8 m wide at the top. The trenches were filled with 15 cm of sand at the bottom, then 30 cm of coarse gravel and 30 cm of pea gravel on the top (Fetter et al., 1976). *Scirpus validus* was used as vegetation cover. Some of the earlier constructed cells contained wire frames to support the emergent wetland plants, but this was quickly abandoned in favor of gravel media. These early gravel-filled marsh cells could be operated with the water above or below the gravel surface, although subsurface flow was identified as the preferred mode of operation. The system was also suggested to upgrade septic tanks effluents.

Biological Water Purification of California, Inc. (BWP) was the California sub-licensee for the Max Planck Institute (MPI) system developed by Dr. Käthe Seidel in Germany (see section 4.3). In May 1976, a joint pilot project was put into operation by BWP of California and the Moulton Niguel Water District. The construction of the trench system was finished by June 1978. The MPI system consisted of the first vertical stage and the second HF stage (both composed of two parallel trenches). Results from the period September 1978 and July 1979 are shown in Table 7-14.

Table 7-14. Treatment performance of the MPI treatment system in Laguna Niguel, California. All values in mg l^{-1}. Data calculated from Pope (1981).

	Inflow	VF out	HF out
BOD_5	189	73	29
COD	407	156	79
TSS	201	45	19
TP	12.8	12.8	12.3
TN	39.6	33.2	26.4
NH_4-N	24.5	20.1	17.6
NO_x-N	0.3	1.5	0.6
$N_{org.}$	14.8	11.6	7.8

During the early 1980s pilot-scale (65 m^2 each) a full-scale (824 m^2 each) HF constructed wetlands were studied at Santee, California (Gersberg et al., 1983, 1984b). In the mid-1980s, Wolverton and co-workers carried out many laboratory and small-scale outdoor experiments with planted filters, called rock/reed filters or microbial-plant filters (e.g., Wolverton, 1982, 1986, 1989; Wolverton and Bounds, 1988; Wolverton et al., 1983, 1984a,b).

During the late 1980s, Tennessee Valley Authority in cooperation with the Kentucky Division of Water, U.S. EPA, and the National Small Flows Clearinghouses, implemented three demonstration projects to investigate and promote the feasibility and benefits of using constructed wetlands for domestic wastewater (Choate et al., 1990; Watson et al., 1990). The Kentucky systems were built at cities of Benton, Hardin and Pembroke. The Benton system (Fig. 7-35) was built in 1986 to upgrade existing lagoon and consists of three cells of equal size (14,652 m^2). Two cells were designed as FWS system, the third cell was a HF wetlands filled with gravel and planted with *Scirpus validus*. The system at Hardin (Fig. 7-35) was built in 1988 to polish the effluent from the package contact stabilization plant. The constructed wetland consisted of two equally sized HF cells (3,190 m^2) planted with *S. validus* and *Phragmites australis*.

The constructed wetland at Pembroke is so called a "Marsh-Pond-Meadow" system. The system consisted of two parallel and independent systems each having in series two parallel "marsh" cells, a pond and a "meadow". Designs of the two systems are the same except for substrate and marsh vegetation. System A was constructed with gravel substrate in the marsh and meadow cells to provide sub-surface flow while System B was constructed with natural clay to provide FWS cells. However, after the wetland system began operating the actual base flow was found to be substantially less than the design flow and it was decided to use only FWS system (Choate et al., 1990). The design and performance of the demonstration systems in Hardin, Pembroke and Benton were well described in the literature (Steiner et al., 1988; Choate et al., 1989, 1990, 1993).

Figure 7-35. Tennessee Valley Authority demonstration HF constructed wetlands at Benton (top) and Hardin (bottom), Kentucky shortly after construction. Photo courtesy Jim Watson.

Another demonstration HF CW project by TVA was completed in August 1988 to polish the effluent from the school's extended aeration package treatment plant at Phillips High School, Bear Creek, Alabama (Watson, 1990). Monitoring results from the period October 1988 through July 1989 revealed that the system was very effective (Table 7-15). The

TVA experience with HF constructed wetlands for small users including individual residences was summarized by Steiner et al. (1991, 1993).

The early experience with HF constructed wetlands in North America was evaluated by Watson et al. (1989), Brown and Reed (1992), Reed and Brown (1992), Knight et al. (1992) or Reed (1993).

During 1990 and 1991 the U.S. Environmental Protection Agency sponsored an effort to identify existing and planned constructed wetlands in the United States (Brown and Reed, 1992). A total of 143 communities with 154 wetlands were identified. The survey identified an almost equal number of operating FWS and HF constructed wetlands but HF systems were projected to outnumber FWS systems in the future. The majority of systems were located in the Mississippi River basin, but they were found or planned throughout the country. HF systems were generally smaller than FWS wetlands and the flow of wastewater ranged between 5 and 11,400 m^3 d^{-1}.

Table 7-15. Treatment performance of HF constructed wetland at Phillips High School, Bear Creek, Alabama during the period October 1988 – July 1989. Data from Watson (1990).

	Inflow (mg l^{-1})			**Outflow (mg l^{-1})**		
	Average	**Min**	**Max**	**Average**	**Min**	**Max**
BOD_5	13	1	30	<1	<1	1.2
TSS	60	40	92	<3	<1	11
NH_4-N	10.7	1.3	20	1.8	0.02	3.6
NO_x-N	26	2.3	49	6.5	0.06	14
$N_{org.}$	12	1.1	28	0.58	0.06	1.4
TN	48	5.7	80	9	1.7	16
TP	6.2	3.4	9	0.29	0.04	0.61
FC (log CFU 100 ml^{-1})	4.19	<2.0	6.02	<1	<1	1

Brown and Reed (1992) pointed out that besides independent designs, there were two major designs found in their inventory. The first one was based on guidance provided by Tennessee Valley Authority (Watson et al., 1990), derived from European experience characterized by the use of small rock or gravel for media (< 40 mm diameter), variable length to width ratios, and lower hydraulic loading rates (< 15 cm d^{-1}). The second design was generally used in EPA's Region IV (South-Central U.S.), derived from work done by Wolverton et al. (1983) characterized by large rock for media (800 mm or greater diameter, large length to width ratios (10 or greater), and higher hydraulic loading rates (> 15 cm d^{-1}).

Knight et al. (1992, 1993) reported on the initiation of North American Database supported by U.S. EPA's Wetlands Research Program. The database included both natural and constructed wetlands for wastewater treatment. Approximately 300 wetland systems had been identified and operational data from 127 systems were entered into the database. However, most data were collected for FWS wetlands and only 15 HF constructed wetlands were included in 1992. North America was slower to adopt subsurface technology as compared to Europe. However, in recent years the use of these systems has drawn more attention and it is estimated that there are about 8,000 subsurface constructed wetlands at present (Kadlec, 2003).

Steiner and Combs (1993) evaluated the treatment efficiency of small on-site HF constructed wetlands treating septic tank effluents in Tennessee, North Carolina and Kentucky. The systems varied between 29 and 258 m^2 and the systems were found to produce a discharge with low BOD_5 and TSS concentrations and also significantly reduce pathogen organisms. Ammonia-nitrogen reduction was variable. However, according to the authors the operators of a small constructed wetlands were pleased with their performance and attractivness.

Neralla et al. (2000) described the performance of eight small constructed wetlands built to improve effluent from septic tanks in Texas (Table 7-16). The wetlands varied in size between 22.5 and 37.8 m^2 and served two to four people. The average HLR was 2.4 cm d^{-1} and varied between 1.8 and 3.0 cm d^{-1}.

Table 7-16 . Average annual performance of eight constructed wetlands treating domestic sewage in Texas. Removal for chemical parameters in %, removal of microbiological parameters in $\log_{10}$. Data elaborated from Neralla et al. (2000).

	Inflow	**Outflow**	**Removal**
BOD_5 (mg l^{-1})	101 (64 – 177)	15 (8 – 22)	84.8
TSS (mg l^{-1})	48 (26 – 114)	8.8 (5 – 16)	76.2
NH_4-N (mg l^{-1})	45 (19 – 84)	27 (11 – 44)	36.2
Soluble P (mg l^{-1})	1.2 (0.6 – 2.2)	0.85 (0.5 – 1.3)	21.0
Total coliforms (log CFU 100 ml^{-1})	6.7 (6.1 – 7.3)	5.3 (5.1 – 5.6)	1.4
Fecal coliforms (log CFU 100 ml^{-1})	6.0 (5.2 – 6.8)	4.4 (3.8 – 5.3)	1.6

Another study aimed at the performance of small single-family constructed wetlands has been reported by Steer et al. (2002). The authors monitored 21 systems in Ohio during the period 1994-2001. Each system had a septic tank for primary treatment and two wetland cells which shared a common design and had an area of 4.5 m (L) x 5.5 m (W), were 0.46 m deep and filled with river gravel cca 30 mm in diameter. One system included in the study was designed with longer. First cells were planted with *Scirpus* or *Sagittaria* (Arrowhead), second cells were planted with ornamental wetland plants such as *Acorus calamus* (Sweet flag), *Lobelia cardinalis* (Cardinal flower), *Asclepsis incarnata* (Swamp milkweed) and *Pontederia cordata* (Pickerelweed). The summary of the monitoring study between September 1994 and January 2001 is presented in Table 7-17.

Table 7-17. Summary of the study on 21 single-family HF constructed wetlands in Ohio, USA. Average values for 3-5 years monitoring periods. Values in mg l^{-1}, fecal coliforms (FC) in CFU 100 ml^{-1}. Elaborated from Steer et al. (2002).

	Inflow	**Outflow**	**Removal (%)**	**EPA limit**
TSS	55.4 (22-163)	18.8 (5-91)	55.8	30
BOD_5	104.7 (10.3-193.3)	13.7 (2.9-48.6)	70.3	30
NH_3	47.7 (6.8-116.8)	18.4 (1.5-46.2)	56.3	1.5
TP	8.4 (4.0-27.4)	1.7 (0.1-16.8)	80.5	1.0
FC	36,410 (730-118,350)	2,150 (50-14,420)	87.9	1000

Chen et al. (1995) described the use of HF constructed wetland for the treatment of dairy wastewater at the University of Southwestern Louisiana. The wastewater was pretreated in anaerobic and facultative lagoons. The authors concluded that HF CWs were more effective than FWS wetlands, especially for removal of TSS.

In the early 2000s, artificially aerated horizontal subsurface flow systems were introduced (Wallace, 2001, Fig. 7-36). There are now several studies which evidenced a positive effect of supplemental aeration (e.g., Drizo et al., 2006).

Figure 7-36. Installation of influent distribution chamber and aeration lines for HF CW at Prinsburg, Minnesota, U.S.A. Photo by Scott Wallace, with permission.

Summary of HF constructed wetlands design guidance (U.S. EPA, 2000) is shown in Table 7-18. HF constructed wetlands are mostly used to treat domestic (Fig. 7-37) or municipal (Fig. 7-38) wastewaters. However, many other types of wastewater such as petrochemical (Wallace, 2002a), food processing (White, 1994; Wallace, 2002b), pulp and paper (Thut, 1990b, 1993), dairy (Hill et al., 2003, Drizo et al., 2006), mining (Pantano et al., 2000) have been treated in HF systems. HF constructed wetlands have also been used for treatment of de-icing fluids from the airport runoff (Karrh et al., 2002) and landfill leachate (Fig. 7-39) (Surface et al., 1993; Bernard and Lauve, 1995; Eckhardt et al., 1999).

Figure 7-37. Single family HF constructed wetland (at the right) in Lindstrom, Minnesota. Photo by Scott Wallace, with permission.

Figure 7-38. HF constructed wetland Morton Farm Preserve, Minnesota for 90 PE. Photo by Scott Wallace, with permission.

Figure 7-39. Pilot-scale (71.7 m^2) HF constructed wetland for landfill leachate at the New Hanover County landfill, North Carolina, U.S.A. Photo by Jan Vymazal.

Table 7-18. Summary of HF constructed wetlands design guidance (U.S. EPA, 2000, with permission).

Pretreatment	**Recommended for use after primary sedimentation (e.g., septic tank, Imhoff tank, primary clarifier). HF CWs are not recommended for use after ponds because of problems with algae**
Surface area	
BOD	6 g m^{-2} d^{-1} to attain 30 mg l^{-1}, 1.6 g m^{-2} d^{-1} to attain 20 mg l^{-1} effluent
TSS	20 g m^{-2} d^{-1} to attain 30 mg l^{-1} effluent
TKN	Use another treatment process in conjunction with HF CW
TP	HF CW is not recommended for phosphorus removal
Depth (typical)	
media	0.5 – 0.6 m
water	0.4 – 0.5 m
Length	Minimum 15 m
Width	Use Darcy's Law; maximum 61 m
Bottom slope	0.5 – 1.0%
Hydraulic conductivity	First 30% of length: 1% of clean k_f Last 70% of length: 10% of clean k_f
Media	All media should be washed clean of fines and debris; more rounded media will generally have more void spaces; media should be resistant to crushing or breakage
Inlet zone	1st 2 meters: 40-80 mm
Treatment zone	20-30 mm, use clean k_f = 100,000, if actual k_f not known
Outlet zone	Last 1 meter: 40-80 mm
Planting media	5-20 mm
Miscellaneous	Use at least 2 HF beds in parallel Use adjustable device with capability to balance flows Use adjustable outlet control device with capability of flood and drain the system

7.3 Central and South America

7.3.1 Brazil

Since 1980, research has been conducted in Brazil on the possibility of the use of water hyacinth ponds in combination with constructed wetlands planted with rice, here called "filtering soil" (Salati, 1987). Under current classification, these systems would be called vertical upflow CWs (for details see Section 4.2.2.2). However, other types of constructed wetlands with emergent macrophytes have been adopted recently (Proceedings, 2000).

Philippi et al. (1999) described a 450 m^2 HF constructed wetland built in 1994 in the Training Centre located in the Agronomica municipal district and belongs to the Santa Caterina State's Research and Technology Company. The system consists of a septic tank and a HF bed. The HF bed (30 x 15 m) is 0.7 deep, filled with mixture of sand, rice hay and clay and planted with *Zizaniopsis bonariensis* (Espadańa).

Meira et al. (2004) tested HF experimental units at the Federal University of Campina Grande, Brazil for treatment of sewage-polluted surface water. The units were planted with rice (*Oryza sativa*) and filled with either gravel or sand. The authors concluded that from a public point of view the effluents could not be reused for unrestricted irrigation but their microbiological features comply with the standards for irrigation of trees, fodder, cereals and animal drinking. Also, the quality of the effluents complies with the Brazilian standards for discharge into surface waters destined for water supply after a conventional treatment.

Philippi et al. (2006) evaluated the treatment performance of four HF constructed wetlands in rural areas under a subtropical climate in Brazil. All systems consisted of a septic tank followed by HF constructed wetland planted with *Zizaniopsis bonariensis*. Bed media is 0.7 m deep and composed of gravel, sand, rice hay and clay. The oldest system – Agronômica – was built in 1994, Videira was built in 2001, Tubãro in 2002 and São Joaquim in 2004. The authors pointed out that treatment efficiency of all systems (Table 7-19) was good and stressed that HF constructed wetlands are easy to operate and maintain and therefore these systems have great potential in rural areas. For systems with low specific area (≤ 1 m^2 PE^{-1}) there was a strong trend of substrate clogging.

Oliveira et al. (2006) reported on the use of HF constructed wetland (64 m^2) as a part of HF-FWS hybrid system to treat wastewater in the Municipal Botanical Garden in the city of Bauru, Brazil. The results from the first 18 months of operation showed very high treatment efficiency already in th HF wetland – 88%, 78%, 79% and 42% for BOD_5, COD, NH_3-N and PO_4-P. The site with the treatment system is used as a community park, and it is used in the application of concepts for the Environmental Education work.

Table 7-19. Removal efficiency (%) of four HF constructed wetlands in Brazil. Inflow and outflow average concentrations (mg l^{-1} for chemical parameters, MPN/100 ml for *E.coli*) are in parentheses. From Philippi et al. (2006), with permission of Portugal Ministry for Environment, Spatial Planning and Regional Development and IWA Specialist Group on the Use of Macrophytes in Water Pollution.

	Agronômica	Videira	Tubãro	São Joaquim
PE*	66	150	50	55
Area (m^2)	450	84	50	40
COD	98% (1,005-19)	69% (317-117)	82% (485-87)	86% (476-68)
BOD_5	98% (979-19)	73% (330-89)	85% (232-35)	80% (201-39)
TSS	53% (224-104)		87% (640-84)	56% (50-22)
NH_4-N	67% (49-16)	56% (34-15)	39% (18-11)	46% (39-21)
TN		-15% (49-56)	68% (48-16)	
PO_4-P	79% (29-6)	81% (27-5)		53% (19-9)
E. coli	2.0 (4.9-2.9)	1.3 (5.9-4.6)	3.4 (7.9-4.5)	0.8 (6.4-5.6)

*number of people connected to the system

7.3.2 Chile

Mariangel and Vidal (2007) pointed out that the constructed wetland technology has not been developed yet in Chile. So far, these systems have only been used or designed in some industrial sectors (slaughterhouse, tobacco processing, wineries, poultry, swine feedlots, kraft mill wastewaters). The systems are mostly FWS-type with few HF applications. Also, 13 rural village wetland systems are of FWS type.

7.3.3 Colombia

Between 1993 and 1999, about 26 HF constructed wetlands were built in Colombia by the company TransForm-Danish Rootzone. The types of wastewater include sewage, compact disc production effluents, oil drilling waters, abattoir, dairy, paper industry, phenolics and heavy metals containing waters and hospital effluents. The systems are usually small with surface area between 12m^2 and 1,200 m^2 (TransForm, 2006).

A two-stage HF constructed wetland with a total surface area of 312 m^2 was built for the Colombian Coffee Growers Federation at the Fundacion Manuel Meija residential training centre (Williams et al., 1999). The system was planted with locally grown *Typha angustifolia*. The whole treatment system including rock filter in gabions for primary treatment and a pond for tertiary treatment proved to be very effective with inflow/outflow concentrations of COD, BOD_5, TSS, phosphate and ammonia-N being 405/32 mg l^{-1}, 360/17 mg l^{-1}, 196/28 mg l^{-1}, 27/7 mg l^{-1} and 7/6 mg l^{-1}, respectively (Williams et al., 1999).

7.3.4 Costa Rica

Dallas et al. (2004) reported the use of HF system for the treatment of greywater in Santa Elena-Monteverde at approximately 1,200 m altitude in the Tilaran mountain range of northwest Costa Rica with a tropical montane climate. The constructed wetland consisted of two beds (17 and 13 m^2) and served four households. The locally available crushed rock (nominal 20 mm) with a porosity of 40% was used to fill the beds. The flow during the monitored period of 12 months was approximately 0.775 m^3 d^{-1}. Due to region's steep topography, wetlands – and native wetland plants – are virtually non-existent. Therefore, *Coix lacryma-jobi* (Job's tears), a wild grass native to tropical Asia and naturalized in Latin America was used as vegetation. The treatment performance of the system is shown in Table 7-20.

Table 7-20. Treatment effect of HF constructed wetland at Santa Elena-Monteverde, Costa Rica. Outflow values are averages for wet and dry seasons. Data from Dallas et al. (2004).

Parameter	**Unit**	**1st bed Inflow**	**1st bed Outflow***	**2nd bed Outflow**
BOD_5	mg l^{-1}	167	8.5	2.0
NH_4-N	mg l^{-1}	8.4	1.0	0.1
TP	mg l^{-1}	1.6	3.6	1.5
PO_4^{3-}	mg l^{-1}	16	12.5	7.0
FC	CFU 100 ml^{-1}	1.5×10^8	18,150	69

Figure 7-40. HF constructed wetland at Santa Elena-Monteverde, Costa Rica planted with a local plant *Coix lacryma-jobi*. Photo by Stewart Dallas, with permission.

Dallas and Ho (2005) tested alternative media for constructed wetlands designed to treat domestic greywaters. They used segments (100-150 mm) of PET plastic drinking water bottles and compared this filtration material with a local crushed rock (20 mm fraction). In both wet and dry seasons, performance of reed beds with PET media for BOD and fecal coliforms was either comparable to or better than that with crushed rock.

7.3.5 Ecuador

In 1999, a HF constructed wetland was put in operation in Shushufindi, Ecuador, to treat wastewater from a slaughterhouse (Lavigne and Jankiewicz, 2000). The system consists of a settling tank and two beds in series with a total surface area of 1,200 m^2, planted with local plants *Echinochloa polystachia* (Caribgrass) and *Panicum maxium* (Saboya). The treatment performance of the system for the period June 199 to January 2000 is shown in Table 7-21. The authors mentioned that a second much larger facility (2 ha) was under construction to treat Shushufindi's 2,000 m^3 d^{-1} of municipal sewage from 10,000 inhabitants.

Table 7-21. Treatment effect of HF constructed wetland at Shushufindi, Ecuador, treating wastewaters from a slaughterhouse. Data calculated from Lavigne and Jankiewicz, 2000).

Parameter	Unit	HF Inflow	HF Outflow
BOD_5	mg l^{-1}	237	4
COD	mg l^{-1}	349	8
TSS	mg l^{-1}	106	1.5
NO_3-N	mg l^{-1}	0.13	1.0
NH_4-N	mg l^{-1}	22.5	4.0
PO_4-P	mg l^{-1}	1.6	0.1
FC	$\log_{10}$ CFU 100 ml^{-1}	5.0	3.3

7.3.6 El Salvador

HF constructed wetlands "San José las Flores" at Chalatenango was put in operation in 2002 (Fig. 7-41). The system surface area is 1,924 m^2 and serves 1,372 PE. The bed is planted with *Pennisetum purpureum* (Elephant grass), a typical pasture plant in Central America which has a high water and nitrogen demand (Platzer et al., 2002).

7.3.7 Honduras

Community HF constructed wetland "Teupasenti" at Danlí, Honduras was built in June 2001 (Platzer et al., 2002). The system serves 2,810 PE and has the surface area of 4,250 m^2. *Pennisetum purpureum* and *Zea mays* (Sweet corn) are used as vegetation cover (Fig. 7-42).

Figure 7-41. HF constructed wetlands “San José las Flores” at Chalatenango, El Salvador. Photo by Michael Platzer, with permission.

Figure 7-42. HF constructed wetland “Taupasenti” at Danlí, Honduras. Photo by Michael Platzer, with permission.

Masi et al. (2006a) reported on the use of small experimental HF constructed wetland (10 m^2) in the city of Nacaome, Department of Valle, to upgrade existing lagoons. The system was established in November 2004 to test the performance before the full-scale system application.

7.3.8 Jamaica

Stewart (2005) described two HF constructed wetlands for septic effluent treatment in Jamaica (Fig. 7-43). The system in Pisgah has two parallel beds (19.1 x 4.7 m), is filled with irregularly shaped washed stone 6-25 mm diameter and is planted with local wild cane (*Gynerium sagittatum*). The HF constructed wetland at Retrieve was designed to treat wastewater from a school. Three parallel cells (14.5 x 2.2 m each) are filled with the same material as in Pisgah and wild cane is used as well.

Figure 7-43. HF constructed wetlands at Pisgah (left, before planting) and Retrive (right, planted with wild cane, *Gynerium sagittatum*) in Jamaica. Photos by Ed Stewart, with permission.

7.3.9 Nicaragua

In Central America, Nicaragua is the country that has least coverage of sanitary installations like sewer systems and centralized wastewater treatment plants, which give service to only 17.3% of the total population (Müller, 1988). Platzer et al. (2002) pointed out that research work on the application of the wetland technology in Nicaragua had begun in 1994. There was an urgent need to find a technically and economically viable alternative for the treatment of wastewater generated by small and medium-size towns in Central America, and which produce effluents with a quality to allow their reuse for irrigation of crops cultivated in the region. The construction of the first Central America full-scale HF constructed wetland in the city of Masaya for 1,000 PE (Fig. 7-44) was financed by Austrian development cooperation funds in 1996.

Based on the results obtained from this plant seven other HF constructed wetlands have been built in Nicaragua since 1999 (Fig. 7-45) and one system was built in Honduras and El Salvador. The systems in Nicaragua usually treat municipal or domestic wastewaters and the size varies between 5 PE and 8,750 PE (Fig. 7-45) with specific area ranging between 0.8 and 1.8 m^2 PE^{-1}. Volcanic gravel is used for filtration beds (Platzer et al., 2002).

Figure 7-44. HF constructed wetland "Villa Bosco Monge" in Masaya, Nicaragua. Total area of the beds is 1,280 m^2. Volcanic gravel (0.5-15 mm) is used as filtration substrate and beds are planted with *Phragmites australis*, *Typha domingensis*, *Cyperus articulatus* and *Pennisetum purpureum*. Photo by Michael Plazter, with permission.

7.3.10 Uruguay

Small HF constructed wetlands planted with *Typha* spp. were tested in Uruguay for the treatment of dairy industry wastewaters (Perdomo et al. (2000). The first full-scale HF constructed wetland was put in operation in 2001 (Fig. 7-46, see also Fig. 5-50). The HF system Nativa I for treatment of the soft drinks production plant effluent consisted of two HF cells planted with *Typha domingensis*, with a total surface area of 1,020 m^2. In 2006, HF system with a total surface area of 1.5 ha and designed flow of 3,000 m^3 d^{-1} was put in operation at San Jacinto meat processing plant (Perdomo, pers. comm., see also Fig. 5-6). For pretreatment screening, settling tanks, anaerobic and facultative anaerobic lagoons are used (Fig. 7-47).

Figure 7-45. HF constructed wetlands “Carretera Sur” at Managua, Nicaragua (5 PE, 7 m^2) planted with *Phalaris arundinacea* (top) and “Chicigalpa” at Chinandega, Nicaragua (8,750 PE, 10,000 m^2) planted with *Phragmites australis* (bottom). Photos by Michael Platzer, with permission.

Figure 7-46. The first full scale HF constructed wetland in Uruguay (NATIVA I) designed for treatment of soft drink plant effluent. Photo by Silvana Perdomo, with permission.

Figure 7-47. Anaerobic (left) and facultative anaerobic (right) lagoons as pretreatment units in HF constructed wetland San Jacinto treating meat processing enterprise wastewaters in Uruguay. For HW beds see Fig. 5-6. Photos by Jan Vymazal.

7.4 Australia, New Zealand and Oceania

7.4.1 Australia

The potential use of aquatic and wetland macrophytes for wastewater treatment was evaluated in Australia by Mitchell during the mid 1970s (Mitchell, 1976). In 1980, the assimilative capacity of wetlands for sewage effluent was evaluated (Bavor et al., 1981) and Finlayson and co-workers carried out pilot-scale experiments on the use of sub-surface constructed wetlands for the treatment of piggery wastes and abattoir wastewater (Finlayson and Chick, 1983; Finlayson et al., 1987). Extensive pilot-scale experiments were also carried out at University of Western Sydney (e.g., Bavor et al., 1987). Davies and Hart (1990a) carried out experiments with artificially-aerated HF constructed wetlands. However, nitrogen removal increased in aerated wetlands only by 22-24%. The early use of wetlands for wastewater treatment in Australia was reviewed by Mitchell et al. (1994).

Greenway and Wooley (1999) pointed out that despite several pilot projects in Australia in the 1980s and early 1990s (e.g., Bavor et al., 1987, 1988, 1989; Finlayson and Chick, 1983; Finlayson et al., 1987; Davies and Hart, 1990a,b; Davies et al., 1993; Thomas et al., 1994), this wastewater-treatment technology has not widely been adopted in Australia (Greenway and Simpson, 1996; Greenway and Woolley, 1999, 2001; QDNR, 2000). Interest in constructed wetlands for the treatment of municipal wastewater diminished in the late 1990s. This may have been because of relatively poor phosphorus removal (Greenway and Woolley, 1999; QDNR, 2000), and government pressure to upgrade and augment sewage treatment plants to produce a very high-quality tertiary effluent, or concern that surface-flow constructed wetlands may be potential breeding sites for mosquitoes (QDNR, 2000). However, the recent study of mosquito larvae in four constructed wetlands in Queensland showed that predation of mosquito larvae by aquatic invertebrates controls the larvae and prevents the development of pupae (Greenway et al., 2003). Thus there are minimal health risks in terms of these wetlands being breeding grounds for mosquitoes, if designed and managed to maximize macro-invertebrate predators. At present, free water surface constructed wetlands in Australia are primarily used to treat stormwater runoff with the aim of producing water for recycling, and particularly for irrigation and recently FWS systems have been again used for tertiary treatment of municipal sewage. Subsurface flow systems have also been popular in some rural parts of the country for in-site treatment of domestic sewage or small enterprises (Fig. 7-48).

Davison et al. (2006) noted that in the early 1990s, a number of studies demonstrated that approximately 40% of domestic on-site wastewater management systems in the coastal zone of New South Wales were failing (Geary, 1992). In response, NSW Department of Local Government introduced a set of guidelines (DLG et al., 1998) which require all local Council in the state to prepare an on-site sewage management strategy with requirements relevant to their particular geographic and climatic conditions. Davison et al. (2006) pointed out that the use of secondary treatment in on-site wastewater management systems is becoming increasingly common in Australia. The relevant standard AS/NZS, 1547 (2000) defines a secondary treatment effluent as one in which TSS and BOD_5 are less than 30 and 20 mg l^{-1}, respectively. Among the number of technologies capable of achieving

Figure 7-48. HF constructed wetland at The Channon treating wastewaters from a laundry. The bed is planted with a mixture of *Typha orientalis* (Cattail) and *Bolboschoenus fluviatilis* (Marsh clubrush). Photo by Jan Vymazal.

these water quality levels with domestic wastewater are also HF constructed wetlands. The authors reported as many as 80 reed beds being approved as on-site treatment system in Lismore City Council during the 5-year period 2001-2005 and Davison et al. (2005) reported that approximately 100 reed beds have been installed in the Lismore City and Byron Shire Council area in the northeastern corner of New South Wales in the last 8 years.

Davison et al. (2005) summarized results from 13 HF constructed wetlands treating greywater, grey- and blackwaters, laundry and school wastewaters. The average removal removals amounted to 81.3%, 82.9%, 56.5% and 34.9% for BOD_5, TSS, TN and TP, respectively. The average removal of fecal coliforms amounted to 1.9 log_{10} units.

Davison et al. (2001) reported on the results of studies on four HF constructed wetlands located in the moist subtropical north eastern corner of the Australian state of New South Wales. All beds were planted with *Phragmites australis*, two systems received wastewaters a school, other two systems treated greywaters. All four units were found to maintain satisfactory treatment performance year round. Mean removal efficiencies ranged from 56 to 90% for TSS, 70 to 93% for BOD_5, 38 to 66% for TN, 42 to 780% for TP and 87 to 99.8% for fecal coliforms.

Headley (2004) and Headley et al. (2001, 2005) reported on the use of HF constructed wetlands for treatment of runoff waters from nursery irrigation.

7.4.2 New Zealand

According to the survey carried out by Sukias et al. (1997a, b), constructed wetlands had been adopted enthusiastically by many New Zealand communities as a cost-effective means of secondary and tertiary wastewater treatment. The survey revealed that there were about 80 constructed wetlands for wastewater treatment excluding those treating stormwaters and farm dairy wastes. Surface flow constructed wetlands (see Fig. 4-28) were most common (45%) followed by subsurface flow (Fig. 7-49) and hybrid systems (35% and 14%, respectively, see also Fig. 4-54). The remaining systems were called “enhanced natural wetlands”. The surface flow systems were much larger (average size 2.2 ha) than those with subsurface flow (average size 0.4 ha). At present, constructed wetlands in New Zealand are also very often used to treat agricultural runoff waters.

van Oostrom and Cooper (1990) used experimental HF wetlands planted with *Scirpus validus* and *Glyceria maxima* to treat partially-treated meat processing effluents with high ammonia concentrations. The authors pointed out that removal of COD and TSS was very good but TN removal was rather low due to limited nitrification.

Tanner (1992) used HF constructed wetlands for treatment of dairy parlor wastewaters on the Ruakura Research Farm near Hamilton. Four beds (9.5 x 2 m each) filled with pre-washed alluvial rhyolitic gravel (10-30 mm diameter, 35-37% porosity) were fed at various hydraulic loading rates for 20 months. The removal of BOD, TSS, TN and TP varied between 70 and 90%, 40 and 90%, 40 and 90% and 30-80%, respectively, in relation to loading rate.

Duncan (1992) described the use of HF constructed wetlands for the treatment of wastewater pretreated in facultative aerated ponds in Coromandel. The total ultimate design area of the beds to provide the future town growth was 11,400 m^2. Two beds with a total area of 3,800 m^2 were initially constructed and commissioned in November 1988 and two further beds of 3,800 m^2 total were commissioned in early 1990. The results from the first year of operation were quite satisfying with average inflow/outflow concentrations of BOD_5: 104/11.5 mg l^{-1} and TSS: 60/11.4 mg l^{-1}.

Tanner et al. (2000) pointed out that although not commonly identified as a specific problem, a number of gravel bed systems with high length to width ratios would have been prone to flooding and overland flow during storm events. These systems had insufficient hydraulic cross-sectional area to retain elevated flows. Such issues can be exacerbated where hydraulic conductivity of the media is reduced by clogging with inorganic sediments

Figure 7-49. HF constructed wetland at Waikeria prison facility, New Zealand. Photo by Jan Vymazal.

(e.g., due to embankment slumping, poor construction practices etc.) or carry-over of organic sludges from preceding treatment stages. Carry-over of sludge during high flow events was a common feature of wetlands linked to aerated lagoons and small package treatment plants. Here the wetland could be seen as providing a valuable buffering role, protecting the receiving water. However, few of these systems appear to have been designed to function in this role.

7.4.3 Fiji

In 2004, a 44 m^2 HF constructed wetland was built in Tangage village, on the Coral Coast of Fiji (Fig. 7-50). The system was designed to treat wastewater from the Chief's residence and village meeting house (15 person equivalents) after pretreatment in a septic tank. The wetland was built by villagers with technical assistance from NIWA, Hamilton, New Zealand and the University of the South Pacific, Suva, Fiji, and funding from the Packard Foundation, USA. The project aimed to demonstrate practical options for improved sanitation and reduction of nitrogen loads to the nutrient-sensitive coral reefs nearby.

Figure 7-50. HF constructed wetland at Tangage, Fiji planted with *Cyperus involucratus*. Photos by Chris Tanner, with permission.

7.5 Africa

7.5.1 Egypt

May et al. (1990), Butler et al. (1990, 1993), Williams et al. (1994) and Stott et al. (1997, 1999) described experiments in the pilot HF (called Gravel Bed Hydroponic) constructed wetlands in Abu Atwa, Ismailia. The HF system under investigation consisted of six inclined channels (50-100 m long, 2 m wide and 0.3 m deep with variable slope between 1:20 in the beginning of the bed to 1:100 at the end of the bed). The beds were filled with gravel and planted with *Phragmites australis*.

Khateeb and Gohary (2002, in Korkusuz, 2005) reported on the pilot scale experimental HF system treating Up-flow Anaerobic Sludge Blanket reactor effluent in Cairo. The authors concluded that HF system planted with *Typha latifolia* performed better than FWS system in terms of COD and TSS but FWS unit removed more ammonia-N.

7.5.2 Kenya

Wastewater from a restaurant and swimming pool resort in Nairobi is treated in a 0.5 ha hybrid HF-FWS constructed wetlands including 1,800 m^2 HF wetland. The HF is filled with 1 meter of gravel covered with 10 cm of soil to support growth of cattail (*Typha* spp.) (Nyakang'o and van Bruggen, 1999).

Removal of phenol from pulp and paper mill wastewaters was studied by Abira et al. (2005) at the premises of the PANPAPER Mills, in Webuye, Kenya. The HF wetland with an area of 30.7 m^2 was filled with gravel to a depth of 0.3 m and planted with *Cyperus immensus*, *Cyperus papyrus, Phragmites mauritianus* and *Typha domingensis*. The inflow phenol concentration varied between 0.43 and 1.7 mg l^{-1} while the outflow phenol concentrations ranged from 0.18 to 0.23 mg l^{-1} and from 0.1 to 0.13 mg l^{-1} for HRT of 5 and 3 days, respectively. However, the outflow phenol concentrations did not consistently meet the set national guideline of 0.05 mg l^{-1} for discharge into surface water in Kenya.

7.5.3 Morocco

Mandi et al. (1996) described the use of HF constructed wetland of a Kickuth type in Marrakesh. The system consisted of three beds (600, 800 and 1000 m^2) filled with clay organic substratum planted with *Phragmites australis*. The reed bed effluent corresponded to B category according to WHO guidelines (WHO, 1989) which could be reused for irrigation of cereal crop, fodder, pasture and trees.

El Hafiane and El Hamouri (2004) used HF constructed wetlands to polish the effluent from high-rate ponds effluents in Rabat. Morocco. The wetlands were effective and the experiments showed that planted units (*Arundo donax*) outperformed unplanted controls by 21-25% for COD, 29-41% for TSS, 14-47% for TP, 35-66% for NH_4-N.

7.5.4 South Africa

The application of constructed wetlands for wastewater treatment in South Africa was initially stimulated by the need to remove residual nutrients from secondary treated domestic and industrial effluents (Batchelor, 2003). Research into the design and application of constructed wetlands suitable for the African situation has been undertaken since the early 1990s (Rogers, 1985; Wrigley and Toerien, 1988; Wood and Hensman, 1989). The early research conducted in South Africa reflected European trends with focus on sub-surface flow systems Alexander, 1987). The results of the early research programes were reported by Wood and Hensman (1989), Batchelor et al. (1990) and Wood and Pybus (1992).

Wrigley and Toerien (1988) reported the use of a pilot HF constructed wetland planted with *Phragmites australis* to upgrade the effluent from a oxidation pond in South Africa. At the HLR of 5.2 cm d^{-1} the water quality of the oxidation pond effluent was upgraded in the HF CW effluent to meet effluent standards of NH_3-N, TSS, *E. coli* and PO_4-P.

The HF system at Grootvlei Power Station in Transvaal is designed to remove nutrients and pathogens from a biofilter plant effluent. Influent total-N, phosphate, COD and fecal coliform concentrations were reduced by 79%, 86%, 67% and 99.9%. A system at Olifantsvlei sewage works near Johannesburg is intermediate between natural wetlands and wetlands constructed with gravel or soil. Three units are geotextile-lined, 120 x 20 x 2 meters.

Reed beds were also established in two rural areas of Gazankulu. At Giyani, population of 90,000, outflow from biological seeping beds is channeled to reed beds, where retention time is four days. Seven beds are alternately isolated and drained down to 10 cm for one day to destroy mosquito eggs. Harvested reeds are used in basket weaving, a traditional industry of the area. A similar system was constructed at Nknowankowa. A 2,000 m^2 HF pilot system was built to treat petrochemical effluents. The beds are filled with waste and coarse ash and planted with *Typha* and *Phragmites* (Wood and Hensman, 1989).

Wood (1990) described additional HF constructed wetlands in South Africa. Several reed bed units were established at Vaal Reefs Gold Mine to polish biofilter effluent phosphate levels prior to discharge to the local catchment, whilst acting as an integral part of a nature reserve within the mine complex, and supplying recycle water to the metallurgical plant. Two beds were constructed using a red silty sand each with a surface are of 1,568 m^2, and four beds with gold slime each with a surface area of 532 m^2. The gold slime is the waste pulverized rock left after the gold has been extracted which represents a considerable waste handling problem for all mining activities. Due to hydraulic permeability constraints the gold slime beds were loaded at 10 cm d^{-1} whilst the red soil beds were loaded at 20 cm d^{-1}. All beds were planted with *Phragmites* sp.

HF constructed wetland at MaKwane treats oxidation pond effluent and consists of two 350 m^2 beds loaded at 10 cm d^{-1}. The system at Moeding is a single bed with an surface area of 990 m^2 was designed to treat oxidation pond effluent at HLR of 20 cm d^{-1}. Both systems are filled with 0.6 m of clayey sand topped with a 0.2 m layer of coarse stone/gravel and planted with locally collected *Phragmites*. The Nkowan Kowa HF constructed wetland consists of six reed beds each of 1,170 m^2 surface area and 0.6 m deep. The beds are operated in parallel with 5 in use at the same time with one rests. The HLR is 37.5 cm^{-1} which provides retention time of 3.9 days. System at a Cotton Gin near Warmbaths consists of 560 m^2 bed filled with a coarse sandy clay soil and planted with *Phragmites* (Wood, 1990).

Treatment performance of 40 m^2 pilot-scale HF constructed wetland operated at HLR of 33 cm d^{-1} and organic load of 1,200 kg COD ha^{-1} d^{-1} was reported by Batchelor and Loots (1996). The average inflow/outflow concentrations during the period 1990/1995 were 390/105 mg l^{-1}, 127/19.2 mg l^{-1} and 26/18 mg l^{-1} for COD, TSS and NH_4-N, respectively. Authors concluded that it is very possible that the HF constructed wetlands can operate successfully at high organic loading rates, however, increase in HLR resulted in surface flow. The authors also suggested that removal processes were predominantly anaerobic – this was supported by high rate of sulfate removal (inflow 53.5 mg l^{-1}, outflow 19.3 mg l^{-1}).

Batchelor (2003) reported on the use of HF constructed wetland at Bethlehem which was designed to upgrade activated sludge/trickling filter effluents. The medium in the wetland consists of <50 mm crushed rock combined with railway ash. Also, the author provided a cost analysis for constructed wetlands. The analysis revealed that when ammonia discharge concentrations lower than 10 mg l^{-1} are required, HF constructed wetlands need to be combined with nitrification column and FWS. At this point, the capital investments are higher that other fixed film (biofilters, RBC) or suspended growth systems (activated sludge). When operating and maintenance costs are factored in, however, the unit treatment cost of wetland system is lower than that of either of the other two largely aerobic systems.

7.5.5 Tanzania

Various pilot-scale HF systems have been studied at the University of Dar es Salaam in order to upgrade effluents from waste stabilization ponds or upflow anaerobic sludge blanket (Mashauri and Kayombo, 2002; Mashauri et al., 2000, Kaseva et al., 2002; Senzia et al., 2003; Kimwaga et al., 2002a,b, 2004; Mwegoha et al., 2002; Mbuligwe, 2004, Table 7-22). Example of a full-scale HF constructed wetland is shown in Figure 7-51.

Table 7-22. Treatment performance of pilot-scale HF constructed wetlands treating anaerobically pretreated wastewater in Dar es Salaam, Tanzania. Values in mg l^{-1}. Data elaborated from Mbuligwe (2004).

	COD	NO_3^--N	NH_4^+-N	PO_4^{3-}-P	SO_4^{2-}
Inflow	117	0.56	24.4	1.95	40.5
Unplanted control	37.5	0.39	8.9	0.91	21.5
Typha latifolia	22.4	0.31	6.1	0.49	11.1
Colocasia esculenta	27.9	0.34	5.9	0.46	8.9

Kaseva et al. (2000) and Kaseva (2004) reported on the performance of pilot-scale HF constructed wetland at the University College of Lands and Architectural Studies near Dar es Salaam to treat effluent from upflow anaerobic sludge blanket unit. The results were quite promising and also indicated higher efficiency of units planted with *Phragmites mauritianus* and *Typha latifolia* over the unplanted unit (see section 5.4.7).

Renalda et al. (2006) reported on the use of experimental HF constructed wetland to remove tannins from the effluent of tannin extracting company in Tanzania. Removal of both COD and tannins was very high suggesting that HF wetlands may be an effective tool for treatment of tannins from wastewater.

7.5.6 Tunisia

The first functional constructed wetland for wastewater treatment in Tunisia was built in 2004 (M'hiri et al., 2005). The system served population of 750 and consisted of primary treatment pond, two VF beds (area of 121 m^2) and HF bed (207 m^2). The wetland was planted with *Phragmites australis* and *Typha latifolia* together with mint (*Mentha* sp.) and filled with gravels with various size. The wetland met easily the Tunisian standards (30 mg l^{-1} BOD^5, 90 mg l^{-1} COD and 30 mg l^{-1} TSS). However it failed to meet very strict standards for TKN (1 mg l^{-1}) and TP (0.05 mg l^{-1}).

7.5.7 Uganda

Okurut (2000, 2001) reported on the use of Kirinya experimental HF system in Jinja to treat pre-settled municipal wastewater. The experimental unit with dimensions of 20 x 16 x 1 m was divided into eight sub-units. Four of these sub-units were planted with *Cyperus papyrus* in floating mode without substrate, two sub-units were planted with *Phragmites mauritianus* in gravel and two sub-units were not planted.

Byekwaso et al. (2002) monitored a FWS-HF constructed wetland treating an effluent from Kasese Cobalt Company's cobalt recovery processing plant (see Fig. 5-5). A 1,000 m^2 HF wetland filled with limestone and planted with *Phragmites mauritianus* received water from a 800 m^2 FWS wetland. The system efficiently removed Pb, Co, Ni, Cu, Cd and Fe. There was no overall removal of zinc and manganese was released form the system. While zinc was released in FWS part and removed in HF part, manganese was released in both FWS and HF parts.

Figure 7-51. HF constructed wetland treating wastewaters from the Ruaha Secondary School in Iringa, Tanzania. Photos by Richard Kimwaga, with permission.

7.6 Asia

7.6.1 China

The first full scale constructed wetland for wastewater treatment in China – Bainikeng Constructed Wetland (Fig. 7-52, Table 7-23) - was put in operation in July 1990 at Longgang, Shenzhen Special Economic Zone (Yang et al., 1994). It consisted of four stages with three stages being HF systems. The first stage (1,512 m^2) was planted with *Phragmites australis* and filled with 3-5 cm gravel, the second stage (1,739 m^2) was planted with *P. australis* and *Cyperus malaccensis* and filled with 1-3 cm gravel, and the fourth stage (2,850 m^2) was planted with *C. malaccensis* and *Lepironia articulata* and filled with 0.5-1 cm gravel. The third stage (1,710 m^2) was a FWS wetlands with floating-leaved and free-floating species (see Fig. 4-11).

Figure 7-52. Constructed wetland at Bainikeng, Longgang, Shenzhen Special Economic Zone, China. The system consists of three HF compartments with a total area of 6,100 m^2 treating 1450 m^3 d^{-1}. Photo by Jan Vymazal.

Table 7-23. Treatment performance of Bainikeng constructed wetland during the period 1991-1993. Data from Yang et al. (1994).

	Inflow	**HF 1 out**	**HF 2 out**	**FWS out**	**HF 3 out**
BOD_5	93	32	17	15	6.9
COD	145	83	60	56	38
TSS	141	35	27	24	11
TN	23.7	22.8	21.9	20.2	18.2
NH_4-N	20.7	19.7	19.1	18.9	18.5
NO_3-N	0.6	0.42	0.35	0.36	0.38
TP	2.3	2.2	1.95	1.75	1.59

Ji et al. (2002) reported the use of HF constructed wetlands to treat heavily oil-contaminated water produced in Liaohe Oilfields. Two wetland cells with the area of 900 m^2 (60 m L x 15 m W) were 0.6 m deep and fed with the water pretreated in waste stabilization pond. The system performed successfully with average outflow concentrations of 104 mg l^{-1}, 4.45 mg l^{-1}, 2.55 mg l^{-1} and 0.1 mg l^{-1} for COD, BOD_5, TKN and TP, respectively. The average treatment efficiencies for COD, BOD_5 and TKN were 74.1%. 86.1%, 78.2%. However, the system did not remove phosphorus. Very efficient was the removal of mineral oils – average inflow concentration of 24.7 mg l^{-1} was reduced to an average concentration of 4.15 mg l^{-1}. All monitored parameters met the discharge standards.

Wang et al. (1994) reported on the use HF constructed wetland for treatment of pig raising farm at Leping, south China. The system consists of screens, sedimentary pond, upflow anaerobic hydrolysis pond, HF constructed wetland and fish pond (For details see section 6.3.1). Junsan et al. (2000) used a 4-stage HF constructed wetland with a total surface area of 449 m^2 for the treatment of pig farm effluents in China (see also section 6.3.1).

In Yantian Industry Area in Baoan District, Shengzhen City, a hybrid constructed wetland (see Fig. 4-47) including two parallel 805 m^2 HF beds as final units was built in 1992 (Wang et al., 1994). At hydraulic loading rate of 36 cm d-1 the BOD_5, COD and TSS inflow/outflow concentrations were 83/59 mg l^{-1}, 194/88 mg l^{-1} and 65/3.2 mg l^{-1}, respectively. Another highly loaded HF constructed wetland (HLR 98 cm d^{-1}) for treatment of municipal sewage was reported in Conghua. The wetland consisted of two cells (20.5 x 12.5 m each) and outflow COD, BOD_5 and TSS concentrations were 9.1, 5.4 and 4.3 mg l^{-1}, respectively.

Zhou et al. (2004) reported on the use of HF constructed wetland to treat agriculture stormwater runoff. Three 10 x 1.5 m beds were filled with gravel (30-50 mm, porosity 43.1%). Two beds were planted with *Phragmites australis* and *Zizania caduciflora*, one bed was left unplanted. The results suggested that at lower loading rates removal of N and P via harvesting is substantial and the significance of nutrient removal decreases with increasing loading rate. The planted beds exhibited substantially better performance than the unplanted bed. For more details see section 5.4.7.

7.6.2 India

Harkare et al. (1996) reported on the pilot scale experiments aimed at the use of HF constructed wetlands for treatment of domestic sewage and also wastewaters from manufacture of ion exchange resins. The authors also reported on the design and construction of three HF constructed wetlands for detergent manufacture wastewaters (200 m^2), and municipal sewage (650 and 600 m^2) in central and south-central India.

Juwarkar et al. (1994) studies a 2700 m^2 HF constructed wetland at Saink School, Bhubaneshwar in Orissa State planted with *Typha latifolia* and *Phragmites karka* Billore et al. (1999) reported the use of 42 m^2 constructed wetland planted with *Phragmites karka* to treat municipal wastewater from a small community of residential areas in Ujjain, Central India. Billore et al. (2006) studied nitrogen removal in a 300 m^2 HF system designed to treat wastewater from the Ravindra Nagar residential colony in Ujjain, central India. During the whole study TKN, ammonium-N, nitrate-N and organic-N were reduced by 59%, 40%, 21% and 86%, respectively. The authors observed a substantial loss of ammonia via volatilization due to very high gravel bed surface (56-62°C) during the summer.

Sundaravadivel and Vigneswaran (2001) recommended the use of HF constructed wetlands for wastewater treatment in semi-urban areas of India. Example of a full-scale HF system is shown in Figure 7-53.

Figure 7-53. A 750 m^2 HF constructed wetland planted with *Phragmites karka* in Ekant Park (Bhopal) built in 2002. Daily flow 80 m^3 d^{-1}. Photo by Suresh Billore, with permission.

Billore et al. (2001) reported on the use of HF constructed wetland to treat the secondary treated distillery effluent from a private distillery, Associated Alcohols and Breweries, Ltd. at Khodigram village in the outskirts of Baraha town in Central India. The treatment system consisted of pretreatment chamber and four cell HF wetland with total area of 364 m^2 planted with *Typha latifolia* and *Phragmites karka* in cells 3 and 4, respectively. For details see section 6.2.8.

7.6.3 Israel

Green et al. (1996) evaluated the performance of 100 m^2 HF constructed wetlands for tertiary treatment of municipal sewage before discharging into the river in Netanya City at the Alexander River Basin. Gravel (0.8 – 2.5 cm) with porosity of 50% was used for the wetlands and the units were planted with *Phragmites australis*, *Typha latifolia* and *Scirpus lacustris*. The results of the experiments are preented in Table 7-24.

Table 7-24. Removal of BOD_5, TSS, TN and TP in tertiary treatment experimental wetlands at the Alexander River Basin, Israel. Data from Green et al. (1996).

	Inflow ($mg\ l^{-1}$)	**Outflow ($mg\ l^{-1}$)**	**Removal (%)**	**Load in ($kg\ ha^{-1}\ d^{-1}$)**	**Load out ($kg\ ha^{-1}\ d^{-1}$)**
BOD_5	94	8.4	91	138	12.3
TSS	96	6.1	94	141	9.0
TN	52	19.7	62	76	29
TP	14.7	7.7	48	21.6	11.3

Avsar et al. (2007) studied in a pilot-scale facility HF constructed wetlands with various filter media (gravel rock, river stone and volcanic tuft) and various plants (Common reed, Sugar cane) to upgrade facultative pond effluent at Sakhin. A combination of *Phragmites australis* and volcanic tufa was found as the most appropriate system.

7.6.4 Japan

Kato et al. (2006) reported on the use of the first full-scale constructed wetland in Japan – a hybrid (VF-VF-HF) constructed wetland for the treatment of milking parlor wastewaters in eastern part of Hokkaido (Fig. 7-54). For HF bed with, surface area of 512 m^2, gravel layer (0.7 m) was topped with 0.2 m of volcanic ash soil with high organic matter content.

7.6.5 Jordan

Al-Omari and Fayyad (2003) used 6 HF constructed wetlands (40 m^2 each) planted with *Sorghum halapense* (Johnson grass) and filled with one layer of sand and two layers gravel of different size to treat municipal wastewater in As-Samra, Jordan. The raw wastewater from As-Samra, which receives about 80% of the wastewater generated in Amman, was treated by an upflow anaerobic sludge blanket reactor. The effluent was then settled for about 10 hours and pumped to two small reservoirs connected in series from where it flew by gravity into constructed wetlands.

In 1999, two HF constructed wetlands were built in Dana (50 m^2) and Azraq (1,000 m^2) for the Royal Society for the Conservation of Nature for sewage treatment (TransForm, 2006).

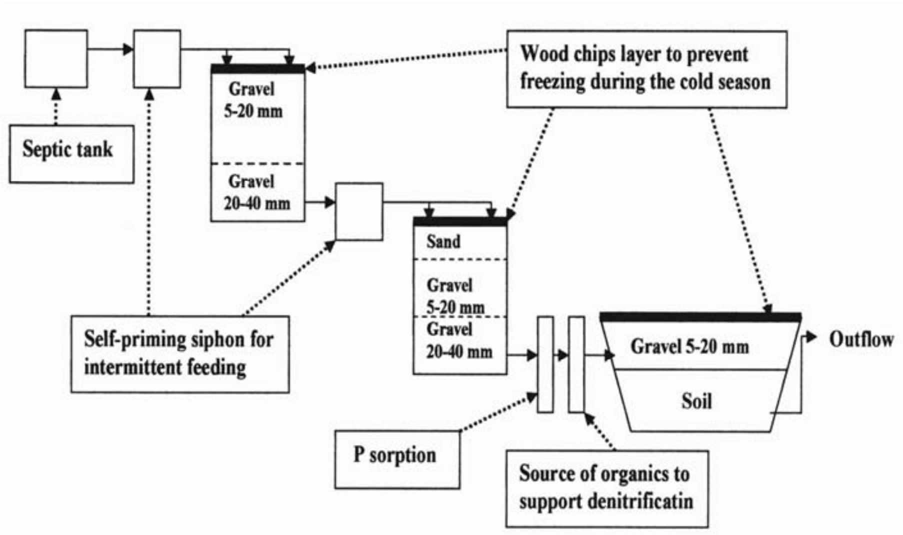


Figure 7-54. The first hybrid wetland system (VF-VF-HF) for the treatment of milking parlor wastewater in Kamishunbetsu, Bekkai-cho, Hokkaido, Japan (working from November 2005). All wetlands are planted with Common reed (*Phragmites australis*). Redrawn from Kato et al. (2006).

7.6.6 Korea

Kwun et al. (2001) tested the use of HSF constructed wetlands for treatment of sewage in order to provide irrigation for paddy rice culture. Yoon et al. (2001) and Ham et al. (2004) reported on the use of HF constructed wetland located at Konkuk University in Seoul, Korea, to test the efficiency of this system as a solution for rural areas in Korea. Ham et al. (2004) found the HF constructed wetland adequate for treating sewage with stable removal efficiency and they recommend the use of polishing pond as a final treatment stage (See also section 5.4.8).

7.6.7 Nepal

HF constructed wetland as the first part of a HF-VF hybrid constructed wetland for hospital wastewater was first used at Dhulikhel to treat wastewater from a hospital (Haberl, 1999; Laber et al., 1999; Shrestha et al., 2001a, Bista et al., 2004). For details see Figure 4-46 and Table 4-20. Shrestha et al. (2001a,b) and Shrestha (2004) reported on the use of a hybrid constructed wetland for Pokhara sub-metropolis consisting of 1,180 m^2 HF bed followed by 2,203 m^2 VF bed designed to treat about 100 m^3 d^{-1} of septage (sludge from a septic tank) and 40 m^3 d^{-1} of landfill leachate. Both HF and VF beds are planted with *Phragmites karka* and HF wetland is filled with gravel 3-6 mm. Another hybrid system (225 m^2 HF-362 m^2 VF) was built to treat septage in Kathmandu City (Shrestha et al., 2001b).

Shrestha et al. (2001a) gave the following observations after three successful constructed wetland applications were built in Nepal:

- CWs appreciated by public, professionals and politicians
- simple construction and operation, socially acceptable
- not necessary to follow exactly the "western" design guidelines to construct the system (design has to be based on local interests, available resources and local climatic conditions)
- use of locally available materials
- low operating costs, no need for qualified personnel for operation
- stable treatment performance, no alteration with the season
- appropriate for establishment of decentralized wastewater treatment plants in urban centers
- more appropriate for the community level where sanitary facilities do not exist
- greywater recycling could be possible.

Bista et al. (2004) reported on HF constructed wetlands at Kathmandu University (216 m^2) and Malpi International School (136 m^2) planted with *Phragmites karka*. The results showed that pollutant removal efficiency of the HF wetlands met the effluent standards recommended by the regulating authorities of Nepal. Khatiwada and Pradhan (2006) described nine constructed wetlands built in Nepal between 1997 and 2004.

Pandey et al. (2006) studied the performance of HF constructed wetland planted with *Phragmites karka* and filled with coarse gravel (grain size of 0.95 and 2.4 mm, porosity 39%). During the study the authors compared planted and unplanted units (see section 5.4.7) and also HF and VF units. The treatment performances of VF and HF planted wetlands were comparable with VF wetland performing better in NH_4-N removal. The overall treatment performance was high and the authors suggested that several small decentralized constructed wetlands built along the river corridor in the Kathmandu valley are expected to be cost effective due to low construction (see section 5.3.1) and operational cost, high performance, and less need to invest in large and long collecting sewers lines.

7.6.8 Oman

The Nimr Reed Bed Project is a project developed by PDO (Petroleum Development Oman), OIT (Oceans Integrated Technologies, United Kingdom) and TransForm-Danish Rootzone, Denmark. The oil in this area of Oman has a very high content of water that causes problems in processing and water disposal. The water after oil separation is highly saline and also contains traces of oil and heavy metals. In 2000, 8,000 m^2 HF constructed wetland was built to treat 3,000 m^3 d^{-1} of water from the oil separation process. The system also includes 24,000 m^2 of planted evapotranspiration ponds in order to reduce large amounts of water (TransForm, 2006).

7.6.9 Taiwan

Jing et al. (2001) described the use of experimental system consisting of FWS and HF constructed wetlands for treatment of highly polluted water from the Erh-Ren River in southern Taiwan which was contaminated with untreated municipal sewage, swine farms runoff and metal-processing factories effluents. The authors concluded that CWs system clearly could be an effective treatment facility for polluted water in natural reservoirs. However, the treatment efficiency was greatly affected by growing season and the presence/absence of macrophytes.

Lin et al. (2005) used HF wetland in combination with FWS wetland to treat wastewater from an intensive shrimp aquaculture under very high hydraulic loading rates (up to 2.7 m d^{-1} for HF stage). The authors concluded that FWS-HF system provided water quality that was suitable for reuse in shrimp aquaculture. Lin et al. (2002) tested FWS-HF system for the treatment of fish aquaculture wastewater with special focus on nutrient removal. The authors concluded that FWS system removed most inorganic nitrogen HF part-removed phosphate and removal of nutrients were sufficient for recycle in the aquaculture system without danger of harming the milkfish (*Chanos chanos*).

Kao et al. (2001) monitored a two-stage (FWS-HF) pilot-scale unit to test removal efficiency for stormwater runoff in Taiwan. The FWS part was planted with *Pistia stratiotes* while the HF part was planted with *Phragmites australis.* Based on the results a field scale wetland (1,200 m^2 surface area) was built inside the campus of National Sun Yat-Sen University to treat a combination of stormwater runoff and untreated wastewaters from the school drains.

Yang and Hu (2005) studied HF mesocosms for treatment of oil-refinery and steel-mill effluents. The mesocosms were filled with gravel and planted with *Phragmites australis* and *Typha orientalis*. The authors concluded that constructed wetlands showed an obvious polishing function for the effluents from oil refining and steel-mill industrial wastewater treatment plants. It was also concluded that for secondary treated oil refining wastewater effluent HF wetland system performed better than FWS system. The authors pointed out that the effluents from a constructed wetland treating industrial wastewaters might be reclaimed and reused for cleaning, cooling and miscellaneous uses, but are not suggested for processing and boiler water purposes.

Lee et al. (2004) used HF CW for the treatment of swine effluent (for more see section 6.3.1).

7.6.10 Thailand

Kantawanichkul et al. (2001, 2003) and Kantawanichkul and Somprasert (2005) reported on the use of HF constructed wetlands as a part of pilot-scale VF-HF system for treatment of pig farm wastewaters in Thailand. For more information see section 6.3.1.

Brix et al. (2006a) reported on the construction of three constructed wetland systems in tsunami-destroyed coastal areas of Thailand. The systems are

- a 220 m^2 HF constructed wetland for the treatment of domestic sewage from the new-established village of Baan Pru Teau in Phang Nga Province;
- a 6,000 m^2 hybrid system (VF-HF-FWS-polishing ponds) with the capacity of 400 m^3 d^{-1} at the island of Phi Phi Don off the west coast of the Thai-Malayan peninsula for municipal wastewater (Fig. 7-55) and;
- about 5,000 m^2 treatment system (HF beds and polishing pond) with about 3,000 m^2 of HF constructed wetlands for treatment of water from polluted rivers Khlong Pak Bang and Khlong Pak Lak at the west coast of Phuket Island.

7.6.11 Turkey

Korkusuz (2005) pointed out that there were no any full-scale constructed wetland applications in Turkey until 2003, except for a few laboratory scale experimental studies (e.g., Ayaz and Akça, 2000, 2001). To foster the practical development of constructed wetlands used for wastewater treatment in Turkey, a pilot-scale hybrid system (Fig. 7-56) was built to treat domestic wastewater produced by 60 PE living in the residential area of the Middle East Technical University (METU) in Ankara (Korkusuz et al., 2002). The system consisted of two parallel VF beds followed by two parallel HF beds with the area of about 90 m^2. Both pairs of beds were filled with gravel/sand and blast furnace granulated slag and planted with *Phragmites australis*. The results were quite promising (Korkusuz et al., 2004).

Yildiz et al. (2004) reported on construction of similar system built in Viransehir, south-east Turkey in early 2004. In this case the configuration was HF (two parallel 150 m^2 beds) –VF (two parallel 75 m^2 beds). The beds are filled with local substrates (basalt and volcanic slag).

Küçük et al. (2003) reported on the use of experimental horizontal sub-surface flow constructed wetland for removal of ammonia from tannery effluents. The bed was filled with layers of gravel, mixture of sand and pea gravel and sand. The area of the bed was 378 m^2. The NH_4-N removal efficiencies of the system at influent NH_4-N concentrations of 10, 20 and 30 mg l^{-1} were approximately 93, 96 and 99%, respectively. The optimum hydraulic retention time was 8 days.

Figure 7-55. Opening day at Phi Phi Don constructed wetland in Thailand. Photo by Hans Brix, with permission.

Figure 7-56. Pilot-scale VF-HF constructed wetland at the Middle East Technical University in Ankara, Turkey. Photo by Asuman Korkusuz, with permission.

Tunçsiper et al. (2006) described the use of a pilot-scale HF constructed wetland (150 m^2) to upgrade effluent from Istanbul Paşaköy Wastewater Treatment Plant. The wetlands was filled with gravel (5 mm diameter, porosity 40%) and planted with *Cyperus* sp.

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SUBJECT INDEX